ASSESSING IMPORTANT BIOLOGICAL FUNCTIONS PROVIDED BY THE WETLANDS RESERVE PROGAM IN OKLAHOMA, USA

By

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Bachelor of Science in Environmental Studies and

Applications

Michigan State University

East Lansing, MI

2007

Submitted to the Faculty of the
Graduate College of the
Oklahoma State University
in partial fulfillment of
the requirements for
the Degree of
MASTER OF SCIENCE
July, 2011

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FUNCTIONS PROVIDED BY THE WETLANDS

RESERVE PROGAM IN OKLAHOMA, USA

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ACKNOWLEDGMENTS

I would like to thank the Oklahoma Agricultural Experiment Station and Oklahoma Cooperative Extension Service through the FY08 Team Initiative Program, the Nebraska Chapter of the Nature Conservancy's J. W. Weaver Competitive Grant, and the Oklahoma State University Graduate College's Summer 2009 Research Fellowship for their funding support. I would also like to thank my committee members, including Craig Davis, Joe Bidwell, and Brian Carter, for their support and guidance. The help of Steve Barner, Ron Goedecke, Nick Jones, and Ed Stinchcomb with the Natural Resources Conservation Service; Bruce Burton with the Oklahoma Department of Wildlife Conservation; and Darren Unruh with the U.S. Fish and Wildlife Service was crucial to locating and providing additional information about study sites. Without the support of my landowners, this project could not have been accomplished. I would also like to thank the City of Chandler with providing accommodations during my field season. The work of Mark Fishbein, who helped with plant identification, and Karen Hickman and Russell Stevens, who assisted with determining plant nativity, was greatly appreciated. For their help with statistical analysis, I would like to thank Dan Storm and Carla Goad. Also, a huge thanks to graduate students who helped with waterbird surveys, including Jesse Burton, Vince Cavalieri, Andy Crosby, Micah Meyer, Adrian Monroe, Jessica O'Connell, Amie Robison, Meredith Roehrs, and Matt Stone. My partner in this

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project, Matt Hough, was a help and support throughout my research. Finally, I'd like to thank all of my friends and family for their support over the years.

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LITERATURE REVIEW

LITERATURE REVIEW

Wetlands are unique and productive ecosystems that provide many important functions. These include hydrologic functions such as storing surface and subsurface water, retaining and slowing floodwaters, recharging and discharging groundwater, and dissipating energy; biogeochemical functions such as nutrient cycling, removing imported elements and compounds, retaining particulates, and exporting organic carbon; and biological functions such as providing vertebrate, invertebrate, and plant habitat (Smith et al. 1995).

Some of these functions are provided by other ecosystems, but wetlands may provide a higher level of function than other ecosystems. For example, at least 33% of all threatened and endangered species in North America live solely in wetlands (Murdock 1994). Also, wetland plant communities are some of the most productive on the planet; they provide habitat and food for many organisms as well as produce significant amounts of organic carbon that can be exported to other systems (Cronk and Fennessy 2001). Functions can also be specific to certain types of wetlands. North Dakota's Devils Lake

Basin prairie pothole wetlands store up to 72% of the total runoff from a 2-year frequency storm and up to 41% from a 100-year storm (Ludden et al. 1983). Playas collect approximately 90% of the runoff from the Southern High Plains region and an estimated 20 – 80% of the water within the playas infiltrates to recharge groundwater aquifers (Haukos and Smith 1994). The prairie pothole wetlands provide breeding habitat for 50% of North America's waterfowl (Smith et al. 1964). Also, over 50% of the species of special concern in Pennsylvania are considered wetland species (Cronk and Fennessy 2001).

Although wetlands provide many important functions, their importance has not always been understood. The historic response to wetlands in the United States during and after European settlement was to fill and drain them for human settlement and conversion to agriculture (Dahl and Allord 1996). This mindset led to significant losses of wetlands across the conterminous United States, as evidenced by Dahl's (1990) estimate that from the 1780s to the 1980s, 53% of the wetlands in the United States were destroyed. The mid-1900s saw a changing attitude toward wetlands in the United States as people began to appreciate the many functions and services they provide. As a result, the average rate of wetland loss in the United States was reduced to 0.05% per year between 1986 and 1997 (Dahl 2000). Moreover, between 1998 and 2004, wetland acreage actually increased 0.03% per year (Dahl 2006). Still, of the original 89.4 million ha of wetlands that occurred in the United States, only 43.9 million ha remain (Dahl 1990, 2006).

Historically, conversion of wetlands to agricultural land accounted for most of the wetland losses in the United States, with the greatest losses occurring in Illinois, Indiana, Iowa, Ohio, and California, where losses ranged from 85% to 91% (Dahl 1990). More recently, most wetland losses are due to urban and rural development, which accounted for 61% of the total losses (Dahl 2006). In addition, wetland losses and gains are not currently equivalent across all wetland types. Based on Dahl's (2006) estimate, a total of 77,630 ha of wetlands were gained between 1998 and 2004. At the same time, approximately 364,540 ha of freshwater shrub wetlands, 57,700 ha of freshwater emergent wetlands, 13,400 ha of estuarine emergent wetlands, and 770 ha of marine intertidal wetlands were destroyed. The offsets of these losses mainly occurred in freshwater non-vegetated wetlands and freshwater forested wetlands, where 289,500 ha and 221,800 ha were gained, respectively. These data show that destroyed wetlands are not being replaced with wetlands of the same type.

Prior to the mid-1900s, federal legislation mainly encouraged draining and filling of wetlands, but legislation such as Section 404 of the 1972 Federal Water Pollution Control Act (also known as the Clean Water Act [CWA]) and the "Swampbuster" provision of the 1985 Food Security Act were passed in the latter part of the 1900s in an attempt to reverse wetland losses in the United States. Section 404 of CWA states that any dredge or fill material in navigable waters of the United States, which was interpreted to include coastal and freshwater wetlands linked to navigable waterways, requires a permit issued by the U.S. Army Corps of Engineers (Cech 2005). Whenever possible, impacts to a wetland must be avoided or minimized. Only if no alternative to draining or destroying the wetland exists will a permit be issued and mitigation may be required to minimize the wetland loss. A permit may be denied if the action produces an unacceptable adverse effect (Cech 2005). The Swampbuster provision denies federal subsidies to farmers that knowingly convert wetlands into farmland or alter wetlands to facilitate cropping elsewhere (U.S. Department of Interior 1994). The Swampbuster provision typically affects areas where farmer participation in government programs is high, but has little effect where farmer participation in government programs is low or where non-program crops predominate (U.S. Department of Interior 1994).

Federal legislation has also authorized the creation of various programs to protect and enhance wetlands across the country. One such program is the Wetlands Reserve Program (WRP), which was authorized by the 1990 Food, Agriculture, Conservation, and Trade Act. WRP is administered by the Natural Resources Conservation Service (NRCS) and was created to provide assistance to landowners who are restoring and protecting wetlands (NRCS 2007a). Current regulations for WRP were enacted by the 1996 Federal Agriculture Improvement and Reform Act, also known as the 1996 Farm Bill. WRP was reauthorized by the 2002 Farm Security and Rural Investment Act, also known as the 2002 Farm Bill (NRCS 2007a).

In 1989, the G. H. W. Bush Administration announced a “no net loss” policy for wetlands, which has been supported by all successive presidential administrations. The goal of the “no net loss” policy is to offset wetland losses by wetland gains in terms of both acreage and, to the extent possible, ecosystem function (U.S. Fish and Wildlife Service 1994). To achieve this goal, any wetland that is destroyed must be replaced with a created or restored wetland (i.e., mitigation wetland) of equal or greater area and comparable functions. To compensate for the lost functions, mitigation wetlands are often larger than the destroyed wetland. Between 1993 and 2000, an average of 1.78 ha was required to replace every hectare of wetland lost (Turner et al. 2001). However, permit requirements are not always met, resulting in fewer acres being provided than are required, and, even when permits are met, the resulting wetlands often do not function as well as the original wetland (Turner et al. 2001).

Wetland creation and restoration are common practices used to meet the goals of the “no net loss” policy as well as to increase functionality on wetlands degraded by factors such as hydrological alterations, salinization, eutrophication, sedimentation, filling, and invasive species (Zedler and Kercher 2005). Created wetlands are those constructed in land that was not previously a wetland. Restored wetlands are those that were degraded, functioning poorly, or no longer present but have been enhanced to help restore them to previous functionality (Mitsch and Gosselink 2007).

Wetland restoration is a complicated task because wetlands are complex systems. Interactions between water, soils, plants, animals, and chemicals all contribute to how wetlands function (Mitsch and Gosselink 2007). Duplicating these intricate wetland interactions is very difficult, which can lead to restored sites that do not match natural systems in terms of structure and function. Two example characteristics of restored wetlands, hydrology and soil, will be discussed to illustrate the difficulty of restoring wetlands. Hydrology is an important factor in wetland function, but the natural hydroperiod of a wetland can vary within a year and between years, making replicating natural hydrologic conditions within restorations challenging. Even a small change in the natural hydroperiod can greatly change a wetland’s functionality (Zedler and Kercher 2005). For example, hydroperiod affects vegetation communities and small differences in a wetland’s hydroperiod can influence which vegetation community predominates, affecting how well a wetland performs the function of “maintaining a characteristic plant community" (Zedler and Kercher 2005, Laubhan and Gleason 2008). Some functions affected by soil such as "maintaining a characteristic plant community" and "groundwater recharge" are changed when soil texture, nutrient status, or microbiota are altered (Zedler and Kercher 2005). Zedler and Kercher (2005) discuss a restored wetland in California that contained a substrate that was too sandy. The soil was too coarse to retain nitrogen, resulting in decreased plant cover. As a result of the decreased plant cover, the population of a scale insect (*Haliaspis spartina*) in the wetland increased and caused further declines in the plant cover because an important predator of the scale insect (*Coleomegilla fuscilabris*) was not attracted to the poor vegetation cover in the wetland. Hydrology and soil characteristics highlight only two of the many interactions within a wetland that can affect restoration success and cause a ripple effect throughout the entire system.

If biotic and abiotic structural components differ between natural and restored wetlands, functions will likely differ as well. Studies comparing functions of created and/or restored wetlands to reference wetlands often show varying results due to differences in created, restored, and reference wetland conditions in different regions and in different wetlands within a region. Studies comparing plant communities between restored and reference wetlands illustrate this. Some studies found similar plant diversity (Hartzell et al. 2007) or richness (Fennessy et al. 2004, Hartzell et al. 2007, Laubhan and Gleason 2008) between created and reference wetlands. Another study found greater plant species richness, evenness, and diversity (Balcombe et al. 2005a) at mitigated sites than reference sites and yet another study showed lower species richness at mitigation sites (Campbell et al. 2002). Balcombe et al. (2005a) argue that the reason for high vegetation species richness in mitigation wetlands was that young created wetlands had a recent disturbance that allowed a wide range of disturbance tolerant species to colonize and species that can competitively exclude pioneer plants did not yet have time to become well established in the wetland. Furthermore, Campbell et al. (2002) suggests that mitigation sites had lower richness because they occurred at greater distances from seed sources than the reference wetlands.

Similarly, avian community comparisons between restored or created wetlands and reference wetlands show varying results. Some studies have shown similar avian species abundance (Balcombe et al. 2005b), richness (Balcombe et al. 2005b, Hartzell et al. 2007) and diversity (Balcombe et al. 2005b, Hartzell et al. 2007) between created or mitigated wetlands and reference wetlands. Desrochers et al. (2008) found a lower breeding bird abundance and richness in created salt marshes than in reference salt marshes, but similar abundance and richness during the non-breeding season. Another study showed created wetlands had a lower density of some avian species and a higher density of other species than natural wetlands and densities per species varied by season (Erwin et al. 1994). When focused only on waterbirds, Balcombe et al. (2005b) found mitigation wetlands had higher waterbird abundance than reference wetlands. Hartzell et al. (2007) reported the proportion of obligate wetland species (e.g., shorebird species, waterfowl species, and rail species) was similar between created and reference wetlands. Balcombe et al. (2005b) attribute higher waterbird abundances in mitigation wetlands to those wetlands containing more open water, less emergent vegetation, and higher plant species richness and diversity than reference wetlands.

Maintaining wetland interspersion, or the ability of a wetland to allow organisms continuous access to food and cover, is another function provided by wetlands. When measured as the wetland density in the landscape, no significant difference existed between mitigation and natural, human impacted wetlands (Kettlewell et al. 2008). When measured as the distance to the nearest wetland, Hoeltje and Cole (2009) found created wetlands to be farther from their nearest neighboring wetland compared to the nearest neighbor distance to reference wetlands because wetlands were created in more fragmented habitats with human disturbance than reference wetlands. However, Lehtinen and Galatowitsch (2001) did not find a significant difference in distance to the nearest wetland between restored and reference wetlands.

If one type of wetland performs better than another type for a function, one cannot assume all functions will be performed better because structural differences in wetland types may affect various functions differently. Created and restored wetlands may perform better than reference wetlands for some functions, but worse or similarly for others. Hoeltje and Cole (2007) found that created wetlands constructed to replace slope wetlands scored higher for functions related to water retention (i.e., energy dissipation/short-term surface water storage, solute adsorption capacity, and retention of particulates) than reference slope wetlands, but scored lower on functions related to maintaining natural conditions (i.e., maintenance of characteristic hydrology, maintenance of native plant community composition and structure, maintenance of characteristic detrital biomass, and maintenance of landscape scale biodiversity) compared to reference wetlands. Created wetlands received a higher score for hydrologic functions because they had lower depth to groundwater, were inundated more often, and had greater depths of standing water, which is a common trend in created wetlands. Created wetlands received a lower score for maintaining natural condition functions because habitats were more fragmented and more disturbed than reference sites.

Wetlands Reserve Program

WRP is a federally administered program to restore, protect, and enhance public and private wetlands and associated uplands in the United States. The goal of WRP is to "achieve the greatest wetland functions and values, along with optimum wildlife habitat, on every acre enrolled in the program" (NRCS 2004, p. 1). NRCS administers WRP and provides technical and financial assistance to landowners who are restoring and protecting wetlands (NRCS 2007a). Other organizations, such as local conservation districts, often work in cooperation with NRCS by providing local outreach and education, identifying priority wetlands, and assisting with developing and implementing conservation planning (NRCS 2007a).

Interested landowners enroll their land for a permanent easement, 30-year easement, or cost-share, depending on how long they choose to enroll and how much financial support they choose to receive. NRCS pays up to 100% of the restoration costs in a permanent easement and up to 75% in a 30-year easement or cost-share (NRCS 2007a). The enrolling landowner must have owned the land for at least one year. Eligible lands include farmed wetlands, previously converted cropland, riparian areas linked to protected wetlands, lands adjacent to protected wetlands that contribute significantly to wetland functions and values, and previously restored wetlands that need long-term protection (NRCS 2007a). Most of the land enrolled is marginal, high-risk, flood-prone agricultural wetlands that have a high potential for restoration (NRCS 2004).

Landowners retain control of the land and are responsible for managing the wetland according to NRCS guidelines. They also control access to the land, but are required to allow NRCS access for monitoring, management, and restoration of the wetland and uplands within the easement boundary (NRCS 2007b). Some rules regarding allowable activities apply as long as the land is under an easement or is included in the cost-share agreement. For example, no permanent buildings can be erected on the site. Some activities, such as haying, grazing, or harvesting timber, require NRCS approval and are only allowed if NRCS determines the activity protects and enhances the purpose for which the easement was acquired (NRCS 2007b).

According to Steve Barner (NRCS, personal communication), after a landowner in Oklahoma enrolls his/her land in WRP, NRCS designs and plans the restoration. The first step is to conduct surveys of the site to determine hydrology, topography, and soil types. A restoration plan is then developed. Plans usually involve at least one dike, a water control structure, and excavations of sloughs and depressions. Some WRP wetlands are more complex and have numerous units, water control structures, excavations, and/or nesting islands for waterfowl. In north-central Oklahoma, most WRP wetlands are dominated by herbaceous vegetation. After construction is completed, trees or native grasses are sometimes planted in the upland to provide a vegetated buffer. The landowner is then responsible for maintaining the wetland by performing tasks such as mowing vegetation, disking soil, and controlling water levels. NRCS is responsible for restoring damaged structures.

Oklahoma NRCS worked closely with Ducks Unlimited (DU) from 2000 to 2008. DU was often responsible for initial site surveys and designing restorations. Oklahoma NRCS's close relationship with DU demonstrates the emphasis of providing waterfowl habitat on WRP sites. This emphasis may cause hydrologic and vegetation management to result in different plant communities, water levels, and hydroperiods in WRP wetlands than reference wetlands in order for the sites to support high waterfowl populations. For example, the water levels of Great Lakes marshes managed for waterfowl were completely different than natural Great Lakes marshes, with low water levels in managed wetlands when natural wetlands were highest and high water levels in managed wetlands coinciding with fall waterfowl migration when natural wetlands were lower (Mitsch and Gosselink 2007).

A main goal of WRP is that the wetlands function as habitat for migratory birds and other wildlife through protection and restoration of wetlands on WRP lands (King et al. 2006) and, more specifically, a goal for many of the WRP wetlands in north-central Oklahoma is to increase habitat for waterfowl (Steve Barner, NRCS, personal communication). Wetlands are particularly important habitat for waterbirds, which depend on wetlands. This dependence makes waterbirds vulnerable to the loss of wetland habitat, and this is particularly a problem in Oklahoma where 67% of the wetlands have been lost (Dahl 1990, Kushlan et al. 2002). However, few published studies have focused specifically on the use of WRP wetlands by wildlife, including waterbirds (Rewa 2005).

No studies have assessed the role of WRP in providing wetland bird habitat in Oklahoma, but some studies have assessed the effectiveness of the WRP and other NRCS habitat restoration programs, such as the Conservation Reserve Program (CRP), in other areas of the United States. In one such study, the waterbird use of restored WRP wetlands and restored reference wetlands in bottomland hardwood forests was compared (Hicks 2003). Species abundance and diversity for waterfowl, shorebirds, wading birds, and marsh birds did not significantly differ between wetland types. However, when date of survey was taken into account, waterfowl were found to be more abundant in WRP wetlands. Kaminski et al. (2006) found waterbird abundance was higher on New York WRP sites with active hydrologic management than on WRP sites without hydrologic management. The increased waterbird abundance was likely due to the larger size of managed sites compared to unmanaged sites as well as the hydrology of managed sites being manipulated to increase the availability of food and emergent cover (Kaminski et al. 2006).

Plant communities play an important role in wetlands by influencing nutrient cycling, hydrology, sedimentation rates, and habitat composition (Mitsch and Gosselink 2007). Because they are closely linked to so many wetland attributes, plants serve as sensitive ecological indicators (Cronk and Fennessey 2001). Plant communities also relate back to NRCS's goal of providing wildlife habitat by providing forage, encouraging invertebrate food sources to become established, and providing cover from predators (Cronk and Fennessey 2001).

Vegetation communities have also been compared between reference and WRP sites. Laubhan and Gleason (2008) compared floristic quality and plant species richness of native and non-native species of wetlands in WRP and CRP grouped by region (Missouri Coteau and Glaciated Plains) and by treatment (cropped, restored WRP and

CRP, and native prairie). Cropped wetlands served as pre-restoration reference wetlands and native prairie wetlands served as post-restoration reference wetlands. Floristic quality was measured using the floristic quality assessment index (FQAI), which assigns ranks to each plant based on the plant's tolerance to disturbance and fidelity, with low values indicating disturbance tolerant species and high values indicating disturbance intolerant species with high fidelity (Andreas and Lichvar 1995). In the Missouri Coteau Region, FQAI of the wetland zone differed between all treatments, with cropped wetlands exhibiting the lowest FQAI and native prairie wetlands exhibiting the highest FQAI. In the Glaciated Plains Region, FQAI was higher in the restored WRP and CRP wetlands than the cropped wetlands, but there was no difference between restored and native prairie wetlands. In the Missouri Coteau Region, the cropped wetlands had the lowest native species richness, while the native species richness in restored and native prairie wetlands was similar to one another. In the Glaciated Plains Region, all treatments had similar native species richness. Regardless of region, non-native plant species richness was similar for all treatments.

Differences in FQAI and native species richness in the Missouri Coteau and Glaciated Plains Regions were likely due to differences in seed bank composition, which contain more wetland species in native prairie wetlands than restored or cropped wetlands, and to differences in hydrologic cycles, which were more dynamic in native prairie wetlands than cropped wetlands. Restored wetlands have been found to have longer hydroperiods and less depth to saturated soil than natural wetlands by other researchers as well (e.g., Cole and Brooks 2000).

Another study compared the ability to provide an environment for the native plant community in restored WRP and CRP wetlands to reference wetlands in four regions (Eckles et al. 2002). They found reference and restored sites in the Prairie Pothole Region provided similar plant habitat, but restored sites appeared to have a significantly lower median for providing an environment for the characteristic plant community than reference sites in Lower Mississippi Alluvial Valley forested wetlands, Central Mississippi Valley forested wetlands, and playa wetlands. Reference wetlands likely performed at a higher capacity due to different hydroperiods and lower sedimentation rates than restored wetlands in all regions where plant community function differed. There was no difference in hydroperiod or sedimentation rates between restored and reference wetlands in prairie pothole wetlands.

Wetland interspersion is an important function provided by wetlands because it helps to provide higher biotic diversity in the landscape than would be provided by more isolated habitats (Brinson et al. 1995). Wetland interspersion promotes biotic diversity by allowing aquatic organisms to immigrate to and emigrate from wetlands as well as allowing terrestrial and aerial organisms to access continuous food and cover (Brinson et al. 1995). It also increases biotic diversity by allowing for the transport of organisms between wetlands on vectors, such as waterbirds transporting eggs and seeds on their feathers and in their digestive tracts (Amezaga et al. 2002). Wetland interspersion also relates to NRCS's goal of providing habitat for birds. When measured as the total wetland area around a wetland, wetland interspersion has been shown to be a significant predictor of bird species richness within 3 km of Prairie Pothole Region wetlands (Fairbairn and Dinsmore 2001) and 10 km of Rainwater Basin playas (Webb et al. 2010).

Only one study was found that assesses WRP wetlands for providing wetland interspersion. In the Prairie Pothole Region, the function "habitat interspersion and connectivity among wetlands" did not appear to be significantly different between restored WRP and CRP wetlands and reference wetlands (Eckles et al. 2002). No reasons were given for why the function was similar between wetland types.

Hydrogeomorphic Approach

An important component of WRP is monitoring and assessment. NRCS recommends using hydrogeomorphic (HGM) functional assessment, a rapid assessment procedure for wetland functions using indirect variables, for areas in which a regional guidebook (explained below) is available (NRCS 2008). The HGM method is composed of two parts: wetland classification and wetland functional assessment.

Classification is a precursor to assessment because assessment requires that all wetlands be compared within the same category. HGM classification relies on three factors to characterize wetlands: geomorphic setting, water source, and hydrodynamics of the wetland (Brinson 1993). Geomorphic setting refers to the wetland's topographic position on the landscape. Water source refers to the where water (e.g., precipitation, surface runoff, groundwater, overbank flow, or tides) within the wetland originates. Hydrodynamics is the flow-direction of the surface or near-surface water flowing into the wetland and the energy of moving water. For example, a riverine wetland's water flows unidirectionally and rapidly moves from overbank flooding of an adjacent river, while a depressional wetland's water flows slowly from water traveling across the surface of the land into the wetland.

HGM classification uses a hierarchical approach to group similar wetlands. The first level of the hierarchy is classes, which includes riverine, depressional, slope, mineral soil flats, organic soil flats, estuarine fringe, and lacustrine fringe, and is based on geomorphic setting, water source, and hydrodynamics (Smith et al. 1995). Classes can be further divided into regional subclasses that are based on additional ecosystem and/or landscape characteristics. Subclass characteristics include climatic regions, dominant plant community, and/or other defining characteristics (Smith et al. 1995). Only after wetlands have been grouped by subclass can comparisons between wetlands be made. Comparing wetlands within a subclass controls variability between wetlands so that any difference between wetland characteristics is due to different levels of functionality and not to inherent differences between wetlands. In essence, wetlands must be compared to similar wetlands.

Hydrogeomorphic assessment is a method designed to rapidly assess wetland functions. The core of HGM assessment is the functional model that is used to determine how well a wetland performs a particular function, also called the functional capacity of the wetland (Smith et al. 1995). A functional model is composed of easily measured conceptual and/or quantitative variables, called functional indices, that contribute to functional capacity. An Assessment Team (A-Team) composed of an interdisciplinary group of scientists creates functional models for a subclass. The A-Team is responsible for classifying wetlands, identifying reference wetlands, constructing functional models, and calibrating the models within a particular subclass (Smith et al. 1995). Results of the

A-Team findings and decisions are published as a guidebook that wetland scientists can use to apply HGM assessment within the specified subclass.

Reference wetlands are used to define the range of conditions, caused by both natural and anthropogenic impacts, for wetlands within a subclass (Smith et al. 1995). Reference standard wetlands are those wetlands that have the highest level of function across the suite of functions (Smith et al. 1995, Smith 2001). These are usually the least altered sites in the least altered landscapes and are used to set functional index conditions. One basic assumption that overarches HGM assessment is that the most sustainable functions are in wetlands that have the fewest human alterations (Brinson 1993, Smith et al. 1995, Smith 2001, Hruby 2001).

Functional indices are a key component to HGM assessment and are used to predict how well a wetland's functions are performing (Smith et al. 1995). A direct measure of functional capacity is the most effective technique, but, often, direct indicators are too difficult to measure or demand too much time to assess. Instead, pertinent structural components of an ecosystem that are necessary for the function to occur are assessed to indicate functional capacity (Brinson et al. 1995). For example, maintaining a characteristic plant community is most accurately measured by a complete survey of the vegetation. However, this process would take several seasons to properly measure, making it costly and time consuming. A model of easily measurable variables can be used instead to estimate the maintenance of the characteristic plant community. Model variables suggested by Brinson et al. (1995) for riverine wetlands include a rapid assessment of the dominant species composition for each vegetation strata, seedling and sapling regeneration, canopy cover, tree density, and tree basal area within the wetland.

Species composition reflects the dominant plant species present; regeneration reflects the continuation of current growth; and canopy cover, tree density, and tree basal area reflect the density of plants. The variables are combined in a model to estimate the functional capacity of the maintenance of characteristic plant community function for a wetland.

OBJECTIVES

WRP Assessment

The goal of WRP is to achieve a net increase in wetland function and acreage in agricultural lands. Recently, NRCS instructed their state offices to assess WRP wetlands to determine if this goal has been achieved (NRCS 2008). Furthermore, NRCS recommended using HGM functional assessment procedures for evaluating WRP wetlands for areas in which a regional guidebook is available (NRCS 2008). As of 2008, a total of 23,620 ha had been enrolled in WRP and 205 WRP contracts had been awarded in Oklahoma, USA. However, no previous studies have been conducted in the state to evaluate if WRP in Oklahoma is restoring and enhancing important biotic wetland functions.

The objective of Chapter 2 was to compare the biotic functions of WRP and natural riverine wetlands along the Deep Fork River in Oklahoma. Specific functions that were compared include (1) maintaining characteristic waterbird communities, (2) maintaining characteristic plant communities, and (3) maintaining wetland interspersion. Because no regional guidebook exists for the study region, HGM assessment could not be used. Instead, waterbird and plant communities were assessed using more direct measures such as relative abundance, richness, evenness, and diversity. Wetland interspersion was assessed using National Wetlands Inventory maps in a Geographic Information Systems program to determine the area of wetlands in the landscape around each study site. Chapter 2 was written as a manuscript for submission to the journal "Wetlands."

Functional Indices Assessment

Validating functional models and their variables should be ongoing during model development (Wakeley and Smith 2001). Model validation for HGM usually entails ensuring an FCI varies sufficiently across the range of conditions. Another approach to model validation is to make certain the indices used in a model do, indeed, relate to the function the model is attempting to measure. Few indices have actually been tested for relevancy to the function and, instead, are chosen based on the best professional judgment of the A-Team (Cole 2006). Without testing indices, there is no way to know whether a model variable relates to wetland function or not (Cole 2006). Knowing which functional indices are correlated with a function provides developers of functional models (i.e., the A-Team) a better idea of which indices should be included in the model because they provide the best correlation with the function.

The objective of Chapter 3 was to determine if relationships exists between a direct measure of the function (i.e., species richness) and potential indirect indices for biotic functions of riverine wetlands in central Oklahoma. The first biotic function assessed for relationship with functional indices was "maintaining characteristic plant communities." The functional indices tested for relationship with plant species richness, which is considered a more direct measure of maintaining characteristic plant communities, were (1) soil organic matter, (2) total nitrogen, (3) available phosphorus, (4) pH, (5) electrical conductivity, (6) median height of the water table, and (7) duration of saturation within the rooting zone. The second biotic function assessed was "maintaining characteristic waterbird communities." The functional indices tested for relationship with waterbird species richness, which is considered a more direct measure of maintaining characteristic waterbird communities, were (1) plant species richness, (2) floristic quality assessment index, (3) vertical structure of the plant community, (4) percent of the plants that were annual, (5) percent of the wetland that was open water or bare ground, (6) duration of inundation, (7) median height of water table, and (8) area of the surrounding landscape that was wetland. Chapter 3 was written as a manuscript for submission to the journal "Wetlands."

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CHAPTER II

ASSESSING IMPORTANT BIOLOGICAL FUNCTIONS PROVIDED BY THE WETLANDS RESERVE PROGRAM IN OKLAHOMA, USA

Abstract: The Wetlands Reserve Program (WRP) is a conservation program administered by the Natural Resources Conservation Service to reverse the loss of wetlands in the United States by protecting, restoring, and enhancing wetlands. To determine if these goals have been achieved, the biotic functions of maintaining (1) waterbird communities, (2) plant communities, and (3) wetland interspersion in WRP wetlands were evaluated to assess whether they are performing similarly to natural wetlands. This research compared eight naturally occurring wetlands to eight WRP wetlands along the Deep Fork River in Oklahoma. Waterbird communities were monitored from June 2009 to May 2010 and plant communities were sampled during the late summer of 2009 and 2010. Waterbird abundance, richness, evenness, and diversity were used to assess waterbird communities. Plant species richness, evenness and diversity were used to assess plant communities. The area of wetland habitat within 3 km of study sites was used to assess wetland interspersion, which is the ability of a wetland to allow organisms continuous access to food and cover. Waterbird community parameters, plant species evenness, and interspersion were similar between WRP wetlands and natural wetlands. However, plant species richness and diversity were

significantly higher in WRP wetlands than natural wetlands. Overall, hydrologically managed WRP wetland restorations along the Deep Fork River are providing similar waterbird communities, plant communities, and wetland interspersion to natural wetlands.

Key words: Oklahoma, waterbird, wetland assessment, wetland function, wetland restoration, Wetlands Reserve Program

INTRODUCTION

Wetlands are unique and productive ecosystems that provide many important functions and values, such as floodwater storage, groundwater recharge, nutrient cycling, pollutant removal, carbon sequesteration, and wildlife habitat (Mitsch and Gosselink 2007). Although wetlands provide important functions, over 50% of wetlands in the United States have been destroyed since 1780 (Dahl 1990). To help prevent the loss of wetlands and their functions, federal legislation over the last 40 years has authorized the creation of various programs to protect and enhance wetlands in the United States. One of these wetland protection laws is the “Swampbuster” provision of the 1985 Food Security Act (i.e., Farm Bill), which gave the Natural Resources Conservation Service (NRCS) a role in wetland protection. The “Swampbuster” provision requires NRCS to deny federal subsidies or other federal payments to farmers that knowingly convert wetlands into farmland or alter wetlands to facilitate cropping elsewhere (U.S. Department of Interior 1994). A later version of the Farm Bill, the 1990 Food, Agriculture, Conservation, and Trade Act, authorized the creation of the Wetlands Reserve Program (WRP). NRCS administers WRP and provides financial and technical assistance to landowners to restore and protect wetlands (NRCS 2007a). Current regulations for WRP were enacted by the 1996 Farm Bill and reauthorized by the 2002 Farm Bill (NRCS 2007a). Since its inception in 1990, over 800,000 ha of wetlands have been enrolled in WRP (NRCS 2008a).

The purpose of WRP is to restore, protect, and enhance public and private wetlands and associated uplands in the United States (NRCS 2007a). The land must be restored to the original natural condition on at least 70% of the project area (NRCS 2004), with emphasis on restoring the physical wetland characteristics (Rewa 2005). NRCS is responsible for restoring enrolled wetlands and repairing water control structures and dikes; landowners are responsible for managing the wetland (e.g., managing water levels and vegetation communities) following NRCS guidelines.

NRCS's goal is to achieve the highest potential wetland functions on WRP sites by restoring wetlands to previous functionality (NRCS 2004). Recently, NRCS instructed their state offices to assess WRP wetlands to determine if this goal is being achieved for physical, chemical, and biological functions (NRCS 2008b). Because WRP wetlands tend to be degraded and are imbedded in an agricultural landscape, the biotic community in them may be severely impacted. Therefore, a major emphasis has been placed on restoring the biotic function of WRP wetlands. However, few studies have been conducted nationally and no studies have been conducted in Oklahoma to evaluate if NRCS is restoring and enhancing important biological wetland functions. Moreover, NRCS is spending considerable amounts of time and money administering WRP (e.g.,

$592.6 million in 2010 [NRCS 2011]), yet the effectiveness of the program in many regions is still unknown. An assessment of biotic functions provided by WRP is important to evaluate the success of restoration. Therefore, the main objective of this study was to evaluate how well WRP wetlands are able to provide three biotic functions: (1) maintaining characteristic waterbird communities, (2) maintaining characteristic plant communities, and (3) maintaining wetland interspersion. Specifically, I compared each of these biotic functions between WRP and natural riverine wetlands along the Deep Fork River in central Oklahoma. These three functions were chosen because direct measures of these functions could be used and they relate closely to NRCS's and landowners' objectives.

A main goal of WRP is to restore habitat for migratory birds and other wildlife through protecting and restoring wetlands on eligible lands (King et al. 2006). In particular, WRP wetlands can provide critical habitat for a myriad of waterbird species (Kushlan et al. 2002). More specifically, a goal of many landowners in Oklahoma is to provide habitat for waterfowl (Steve Barner, personal communication). The importance of WRP to waterbirds is evident by the North American Waterbird Conservation Plan, which recommends using WRP to benefit and provide habitat for waterbirds (Kushlan et al. 2002). Although WRP certainly can benefit waterbirds, few published studies have focused specifically on their use of WRP wetlands (Rewa 2005).

Maintaining characteristic plant communities in WRP wetlands is very important because of the important role the plant community plays in influencing nutrient cycling, hydrology, sedimentation rates, and habitat composition of the wetland (Mitsch and Gosselink 2007). Furthermore, because wetland plants are closely linked to so many wetland processes and functions, wetland plants can serve as ecological indicators of a wetland's condition (Cronk and Fennessey 2001). Plant communities also relate back to NRCS's goal of providing wildlife habitat because of the role the plant community plays in providing food for wetland wildlife, enhancing invertebrate communities, and providing concealment cover from predators (Cronk and Fennessey 2001). Therefore, by monitoring the plant communities of WRP wetlands, we can assess their condition and better understand how well these wetlands are functioning in providing habitat for wildlife species.

Wetland interspersion is the ability of a wetland to allow organisms access to continuous food and cover (Brinson et al. 1995). It is an important function provided by WRP because it helps to provide higher biotic diversity in the landscape than would be provided by more isolated habitats (Brinson et al. 1995). Wetland interspersion promotes biotic diversity by not only allowing organisms to access continuous food and cover through habitat corridors (Brinson et al. 1995), but also allowing for the transport of organisms between wetlands on vectors, such as waterbirds transporting eggs and seeds on their feathers and in their digestive tracts (Amezaga et al. 2002). Wetland interspersion is also critical to waterbirds. Bird species richness was higher on wetland complexes composed of smaller wetlands than on larger, more isolated wetlands (Brown and Dinsmore 1986). Wetland interspersion measured as the total wetland area in the surrounding landscape of a wetland has been shown to be a significant predictor of bird species richness within 3 km of wetlands in the Prairie Pothole Region (Fairbairn and Dinsmore 2001) and 10 km of playas in the Rainwater Basin Region (Webb et al. 2010). Although WRP wetlands may enhance the wetland interspersion function of a landscape and result in increased biotic diversity, NRCS generally has not focused on monitoring this important function.

METHODS

Study Area

Study sites were located along the Deep Fork River in central Oklahoma. I evaluated 16 wetlands that included eight WRP wetlands and eight natural wetlands that were located in Lincoln, Creek, Okfuskee, and Okmulgee Counties within the Cross Timbers and Cherokee Prairie Major Land Resource Areas (Soil Conservation Service 1979). All of the study wetlands were herbaceous, riverine wetlands that received overbank flow from the Deep Fork River, which typically occurs at least once every five years. As of 2009, WRP wetlands ranged in age from 3 to 12 years since restoration with an average age of 7 years since restoration (Table 1). The average size of WRP wetlands was 28 ha (range of sizes: 1-91 ha). Types of restoration techniques used on WRP wetlands included construction of at least one dike, insertion of at least one water control structure, and excavation of some depressions and sloughs. WRP wetlands contained 1 to 4 management units that were separated by dikes. According to NRCS personnel (Steve Barner, Ron Goedecke, Nick Jones, Ed Stinchcomb; NRCS; personnel communication), all WRP wetlands were actively managed using water control structures and some of the sites were also managed by pumping water from the Deep Fork River. Sites also received natural inflow of water from overbank flow and upland runoff.

Natural wetlands ranged in size from 1 to 20 ha, with an average size of 11 ha (Table 1). Natural wetlands were not hydrologically managed and only received water from natural overbank flow and upland runoff.

The region has warm, humid summers and mild winters. The mean annual temperature is 15°C, with highest mean temperatures occurring in July (27°C) and lowest mean temperatures occurring in January (2.1°C; Oklahoma Climatological Survey 2001). The mean annual precipitation is 107 cm, with the highest mean precipitation occurring in May (14 cm) and the lowest mean precipitation occurring in January (4 cm; Oklahoma Climatological Survey 2001). Along the Deep Fork River, the major soil types are Eufaula-Dougherty-Konawa, Osage-Verdigris, and Stephenville-Darnell-Niotaze (Carter and Gregory 2008). All wetland sites are underlain by frequently or occasionally flooded soils (NRCS 2007b). Sites range in elevation between 190 m and 265 m above sea level with a mean elevation of 235 m (NRCS 2001). Because these wetlands were within the floodplain, little elevational differences exist between the river and the wetlands.

Waterbird Surveys

I surveyed waterbirds during the breeding season (June – early July 2009), fall migration (mid-August – early December 2009), and spring migration (March – early May 2010). I considered any species that used wetlands for a portion of their life cycle as a waterbird species. These species included all waterfowl, wading birds, and shorebirds as well as some wetland passerines (hereafter, referred to as "passerines") such as red-winged blackbird (*Agelaius phoeniceus*) and common yellowthroat (*Geothlypis trichas*) and other wetland species (hereafter, referred to as "other species") such as American coot (*Fulica americana*) and belted kingfisher (*Ceryle alcyon*). I used unlimited sight distance point counts with no overlap of point plots to survey waterbirds (Hartzell et al. 2007). Points were systematically located throughout each wetland during site visits to allow for maximum visibility and coverage of the wetland. The number of points per wetland was determined on-site and varied between two and four points per unit, depending on the size and visibility distance of the wetland. Waterbird surveys were conducted in all units. If the vegetation surrounding a point was too dense to observe birds, I walked transects between points to flush birds from cover.

During the breeding season, each wetland was visited twice. Surveys were conducted between sunrise and 4 hr after sunrise (Ribic et al. 1999). At each point, all birds seen or heard were recorded for 10 minutes. At the end of each observation, I used playback calls to determine presence of secretive species, such as rails and bitterns, following the protocols of Ribic et al. (1999). During fall migration, each wetland was visited twice during shorebird migration (late August – September 2009) and twice during waterfowl migration (late October – December 2009). During spring migration, each wetland was visited once during waterfowl migration (March 2010) and once during shorebird migration (mid-April – early May 2010). During fall and spring migration, I recorded all birds seen or heard at each point for 10 minutes. Surveys occurred during daylight hours. During all waterbird survey periods, a 10-day minimum was set between site visits (Desrochers et al. 2008). Birds flying over the wetland, but not landing in it, were not recorded (Best et al. 1998). Surveys did not occur when winds exceeded 25 km/hr or during precipitation events (U.S. Environmental Protection Agency 2002).

Vegetation Surveys

Vegetation surveys were conducted from July to October 2009 and August to September 2010. I conducted vegetation surveys along two transects that were located within each natural wetland or each management unit within each WRP wetland. Transects were situated perpendicular to the elevational gradient in each wetland and traversed the entire width of the wetland. Occurrence of plant species in wetlands was determined using the step-point method for which each plant species is recorded approximately every 1 m (Bonham 1989, Smith and Haukos 2002). Any plant not identifiable in the field was collected and returned to the lab for identification. Plants were identified to the lowest taxonomic level using Tyrl et al. (2007) and Mohlenbrock (2005, 2006, 2008, 2010). Nomenclature is based on the U.S. Department of Agriculture's PLANTS database (U.S. Department of Agriculture 2011).

To assess the plant community in terms of providing habitat to waterbirds, I measured the vertical structure of the plant community (represented as visual obstruction) by using a Robel pole (Robel et al. 1970). Visual obstruction was recorded at 6 randomly selected points located throughout each wetland during September and October 2009. At each point, I recorded visual obstruction from the four cardinal directions by standing at a 4 m distance from the pole at a height of 1 m. Points were randomly selected using Minnesota Department of Natural Resources (DNR) Random Sample Generator version 2.2 (Minnesota DNR, St. Paul, Minnesota, USA) in ArcView version 3.3 (ESRI, Redlands, California, USA).

Interspersion of Wetlands

The interspersion of wetlands was indirectly measured as the percent of surrounding landscape that was wetland. I used National Wetlands Inventory (NWI) maps in ArcMap version 10 (ESRI, Redlands, California, USA) to determine the position of wetlands in the area (Fairbairn and Dinsmore 2001, Cunningham et al. 2007). Any open water observed on digital ortho imagery layers from the National Agriculture Imagery Program (Farm Service Agency, Aerial Photography Field Office, Salt Lake City, Utah, USA) was traced and then added to the NWI coverage. To determine the amount of wetland area surrounding each study wetland, I created a 3 km buffer around each study wetland. I selected the 3 km buffer because past research has shown that the amount of wetland area within 3 km of a wetland is a significant predictor of waterbird species richness (Fairbairn and Dinsmore 2001). The percent of wetland area was calculated as the percent of the 3 km buffer that occupied by wetlands on the corrected NWI maps.

Data Analyses

For waterbird and plant data, species richness was calculated as the total number of species observed in each wetland. Evenness was estimated using Simpson's Reciprocal Index and diversity was calculated using the Shannon Index (Magurran 2004). Relative abundance was only determined for waterbirds and was calculated as the total number of waterbirds observed. The similarity of waterbird assemblages and similarity of plant assemblages between treatments was measured using Jaccard's community similarity coefficient (Krebs 1989). From the waterbird community data, I calculated the proportion of waterbirds that were waterfowl, shorebirds, wading birds, passerines, or other species. From the plant community data, I determined the proportion of plants that were non-native species, perennial species, annual species, obligate wetland species (OBL; >99% occurrence in wetlands), facultative wetland species (FACW; 67 – 99%), facultative species (FAC; 34 – 66%), facultative upland species (FACU; 1 -33%), and obligate upland species (UPL; <1%). I used the PLANTS Database (U.S. Department of Agriculture 2011), the Oklahoma Biological Survey website (Hoagland 2004), and Tyrl et al. (2008) to determine species' nativity. The PLANTS Database was used to determine wetland indicator status and whether a species was annual or perennial. Because some plants (e.g., balloon vine [*Cardiospermum halicacabum* L.], redroot flatsedge [*Cyperus erythrorhizos* Muhl.]) occur as both perennials and annuals depending on environmental conditions, I included these species in both perennial and annual counts, resulting in the sum of perennial and annual proportions exceeding 100%.

The floristic quality assessment index (FQAI) was used to reflect a site's level of disturbance, with low FQAI values indicating higher disturbance than high FQAI values (Matthews et al. 2005). FQAI was assessed to measure how the plant communities in WRP and natural wetlands have been influenced by disturbance. I calculated FQAI by assigning each plant species a coefficient of conservatism (Andreas and Lichvar 1995). The coefficient ranges from 0 to 10 and is based on the plant's nativity and disturbance tolerance. A rank of 0 indicates an invasive plant, a 1 indicates a native plant with high tolerance and low fidelity, and a 10 indicates a native plant with low tolerance and high fidelity (Andreas and Lichvar 1995). I determined coefficients based on values reported by Andreas and Lichvar (1995) and Hartzell et al. (2007).

I analyzed all data using MINITAB version 16.1 (Minitab Inc., State College, Pennsylvania, USA). Prior to conducting analyses, waterbird and plant abundance, richness, evenness, and diversity data as well as wetland area were log transformed in order for the species-area relationship to be linear (Palmer et al. 2008). They were then tested for normality using an Anderson-Darling test and for equal variance using an F-test (Minitab Inc. 2010). Any data that were not normally distributed were rank transformed (Conover and Iman 1981). I used a two-way analysis of covariance (ANCOVA), with log of wetland area as the covariate to examine differences in each waterbird and vegetation metric between WRP and natural wetlands (Matthews et al. 2009). Relative abundance, species richness, evenness, and diversity were the dependent factor in each separate ANCOVA. Wetland type, season, and interactions were the independent factors for bird metric analyses, and wetland type, year, and interactions were the independent factors for vegetation metric analyses. I set wetland area as the covariate because the number of waterbird and plant species in a wetland increases as wetland area increases (Houlahan et al. 2006, Smith and Chow-Fraser 2010).

Prior to analysis, proportion data for waterbirds and plants, FQAI, visual obstruction, and percent of surrounding landscape that was wetland were tested for normality using an Anderson-Darling test and for equal variance using an F-test (Minitab Inc. 2010). If data were not normally distributed, they were square root, cube root, or square transformed (Helsel and Hirsch 2002). For data that were not normally distributed after transformations, I used a Mann-Whitney test (Conover and Iman 1981). Normally distributed data were analyzed using a two-sample t-test. An alpha level of 0.05 was used for all statistical tests.

RESULTS

Waterbird Community

Overall, a total of 46 waterbird species were observed during the study (see Appendix 1 for a complete species list). In WRP wetlands, I observed 43 waterbird species overall, 25 during the breeding season, 31 during fall migration, and 26 during spring migration (Table 2). The most common group in WRP sites overall and during fall and spring migration was waterfowl, which comprised 51%, 61%, and 44% of waterbirds observed, respectively. During the breeding season, passerine was the most common group observed (42%). A total of 11 species (American white pelican [*Pelecanus erythrorhynchos*], Baird's sandpiper [*Calidris bairdii*], black-bellied plover [*Pluvialis squatarola*], greater scaup [*Aythya marila*], gull spp. [*Larus* spp.], lesser yellowlegs [*Tringa flavipes*], pectoral sandpiper [*C. melanotos*], redhead [*Aythya americana*], sora [*Porzana carolina*], spotted sandpiper [*Actitis macularia*], and Virginia rail [*Rallus limicola*]) were observed solely in WRP wetlands. I observed 35 species in natural wetlands overall, and 13, 23, and19 species during the breeding season, fall migration, and spring migration, respectively. Waterfowl was the most common group overall (60% of waterbirds observed), during fall migration (61%), and during spring migration (83%). Passerine was the most common group during the breeding season, comprising 48% of individuals observed. Only 3 species (bufflehead [*Bucephala albeola*], marsh wren [*Cistothorus palustris*], and ring-necked duck [*Aythya collaris*]) were observed solely in natural wetlands.

The overall similarity of waterbird assemblages between WRP and natural wetlands was moderate (J = 68.1%). Similarity between waterbird assemblages in WRP and natural wetlands was reduced when each season was examined separately. Similarity of waterbird assemblages between WRP and natural wetlands was 56.0%, 54.3%, and 55.2% during the breeding season, fall migration, and spring migration, respectively.

Overall, waterbird abundance, richness, evenness, and diversity were not significantly different between WRP and natural wetlands (Table 3). Wetland area had a significant positive effect on waterbird abundance ($F_{1, 47} = 57.68$, $P < 0.001$), richness ($F_{1, 47} = 35.21$, $P < 0.001$), evenness ($F_{1, 47} = 9.27$, $P = 0.005$), and diversity ($F_{1, 47} = 7.37$, $P = 0.011$). Neither season ($F_{2, 47} \leq 0.37$, $P \geq 0.690$) nor any interactions ($F \leq 1.74$, $P \geq 0.195$) had a significant effect for any of the metrics. Overall, natural and WRP wetlands had similar proportions of waterfowl, wading birds, and other species, but natural sites exhibited a larger proportion of passerines and smaller proportion of shorebirds than WRP sites (Table 4).

Vegetation Community

A total of 118 plant species were observed in the study wetlands (see Appendix 1 for a complete species list). In WRP wetlands, I observed a total of 99 species and 52 species were solely in that treatment, the most common of which were giant ragweed (*Ambrosia trifida* L.), green foxtail (*Setaria viridis* (L.) P. Beauv.), and upright burhead (*Echinodorus berteroi* (Spreng.) Fassett; Table 5). In natural wetlands, 66 plant species were observed. A total of 19 species were observed solely in natural sites; Carolina mosquitofern (*Azolla caroliniana* Willd.), Columbian watermeal (*Wolffia columbiana* Karst.), and saltcedar (*Tamarix ramosissima* Ledeb.) were the most common of these. The Jaccard's similarity coefficient between WRP and natural sites for plant assemblages was low (39.8%).

Plant species richness and diversity were significantly higher in WRP than natural wetlands, but evenness did not differ between the two (Table 6). Wetland area had a significant positive effect on plant species richness ($F_{1, 31} = 20.99$, $P < 0.001$), evenness ($F_{1, 31} = 4.30$, $P = 0.049$), and diversity ($F_{1, 31} = 4.48$, $P = 0.045$). Neither year ($F_{1, 31} \leq 0.28$, $P \geq 0.601$) nor any interactions ($F_{1, 31} \leq 1.01$, $P \geq 0.325$) were significant for any of the metrics. Vertical structure (df = 14, t = 2.40, P = 0.031) and FQAI (df = 14, t = 3.69, P = 0.002) were significantly higher in natural than WRP wetlands. Natural wetlands contained a larger proportion of perennial plants and a smaller proportion of annual plants than WRP wetlands (Table 7). Natural and WRP wetlands contained the same proportion of non-native, FACW, FAC, FACU, and UPL plants, but natural wetlands contained a larger proportion of OBL plants than WRP wetlands.

Interspersion

The percent of the surrounding area that was wetland did not significantly differ between natural and WRP wetlands (df = 1, W = 67.0, P = 0.958). Natural wetlands had a mean of 15.8% (±3.0 SE) wetland area within a 3 km buffer, while WRP wetlands had a mean of 18.4% (±3.2 SE). The majority of wetlands within the 3 km buffers was in the riverine class. The only other type of wetland class present was depressional, which was mostly composed of farm ponds. Based on the flooding frequency of soils, I estimated that approximately 70% of wetlands within the buffers were riverine and 30% were depressional.

DISCUSSION

Waterbirds

Waterbird abundance, richness, evenness, and diversity were similar between natural and WRP wetlands. Although differences in mean abundance, richness, evenness, and diversity exist, they were largely explained by wetland area, which had a significant effect on all metrics, with natural wetlands having a lower mean area (11 ha) than WRP wetlands (28 ha). Other studies have shown a similar relationship between restored/created and natural wetlands, with waterbird abundance (Brown and Smith 1998, Hicks 2003), richness (Brown and Smith 1998, Balcombe et al. 2005a), and diversity (Hicks 2003) not differing between restored or created and reference wetlands. Balcombe et al. (2005a) attributed similar bird species richness in part to similar landscape position between reference and created wetlands, which may also be the case in my study. Surrounding land uses, such as the amount of wetland, grassland, and cropland, have been shown to influence abundance of waterbird species and groups (Fairbairn and Dinsmore 2001, Naugle et al. 2001, Webb et al. 2010). For example, waterbirds may be attracted to wetlands near cropland because they can use cropland as a source of food. Also, Fairbairn and Dinsmore (2001) suggest that waterbirds are initially more attracted to larger wetland complexes than isolated wetlands. Based on similar interspersion measurements for my wetlands and personal observations, the amount of wetlands and land uses surrounding WRP and natural sites in this study appeared to be similar, which may have also contributed to the similarities in waterbird communities between natural and WRP wetlands.

The proportions of individuals in the passerine and shorebird groups were the only two groups to differ between wetland types. The proportion of passerine species was higher in natural than WRP wetlands, which is likely due to a greater abundance of large perennial plants, such as cattails (*Typha* spp.) and giant cutgrass (*Zizaniopsis miliacea* (Michx.) Döll & Asch.), in natural sites. Red-winged blackbirds, which make up most of the passerines found in wetlands, are generalist species for which cattails provide suitable habitat (Maddox and Wiedenmann 2005). However, these tall, dense species provide less suitable habitat for other waterbird species such as waterfowl, especially during migration, because they prefer open water for feeding (Payne 1998, Ross and Murkin 2009). In contrast, the proportion of shorebirds was higher in WRP than natural wetlands. Because shorebirds prefer sparsely vegetated habitats over more densely vegetated ones (Davis and Smith 1998), they likely prefer WRP over natural wetlands because WRP sites contained less vegetation cover, especially dense cover, than natural wetlands. Moreover, more exposed mudflats, a preferred habitat of shorebirds, occurred in WRP than natural wetlands, which exhibited 29% and 9% bare ground and open water during vegetation surveys, respectively.

Waterbird assemblages were moderately similar between WRP and natural wetlands. Jaccard's similarity coefficient was 68.1% over the course of the entire study period, which was similar to the 65% avian community similarity observed in created and natural depressional wetlands in central Oklahoma by Hartzell et al. (2007). However, my similarity dropped to 54 – 56% when calculated for each season. Although waterbird species richness, evenness, and diversity were similar between WRP and natural wetlands, Jaccard's similarity coefficient shows that some differences in the composition of species exist between wetland types. Several reasons may be contributing to the differences in waterbird assemblages. For example, the different amounts of large perennial species and open area discussed above may be having an impact. Maximum water depth in wetlands can also affect waterbird composition. For example, occurrence of diving ducks was shown to be positively related with maximum water depth while total species richness was negatively related with maximum water depth (Webb et al. 2010). In a companion study of the same wetlands as my study, Hough (2011) showed that the maximum annual water depth was significantly higher in WRP wetlands (61.6 cm) than natural wetlands (37.5 cm), which may be affecting waterbird composition.

Vegetation

Plant species richness and diversity were both larger in WRP than natural wetlands. Although some of the WRP wetlands in this study were farmed prior to enrollment in the program, seed banks in farmed wetlands have been shown to recover after restoration (Middleton 2003). Farmed wetlands still contain seeds of wetland plants, particularly herbaceous species, that can revegetate after wetland restoration (Middleton 2003). As plant surveys of my wetlands demonstrated, wetland plants have had an opportunity to re-establish in WRP wetlands. Because of this, plant richness and diversity were likely higher in WRP sites due to differences in water table levels and less so due insufficient time for wetland seed banks to recover in WRP wetlands.

Differing water levels between wetland types can explain the higher plant richness and diversity as well as the lower proportion of OBL species in WRP wetlands than in natural wetlands. Shorter flooding duration has been shown to increase plant species richness (Casanova and Brock 2000). Longer flooding events allow only water tolerant species to germinate, while shorter flooding events allow time for terrestrial seeds to also germinate, causing a greater species richness and lower proportion of OBL species when flooding duration is shorter (Casanova and Brock 2000). Hough (2011) determined that the water table was in the rooting zone of the soil or above the soil surface (i.e., 30 cm below the soil surface or above) for a significantly greater portion of the growing season in natural wetlands (81.9%) than WRP wetlands (69.0%) due to use of water control structures in WRP wetlands. The shorter duration of saturation in WRP wetlands has allowed a wider range of species to germinate and grow in WRP wetlands than natural wetlands. The large majority (81%) of species was OBL in natural wetlands because duration of saturation was likely too long to allow most other species to germinate, whereas only 52% of species were OBL in WRP wetlands.

Management practices of WRP sites can explain the lower FQAI, lower visual obstruction, lower proportion of perennial plants, and higher proportion of annual plants than in natural wetlands. Based on personal observation and NRCS personnel (Steve Barner, personal communication), moist-soil management was used by most WRP landowners to increase waterfowl use of their wetlands. This strategy incorporates the use of water level management using water control structures and direct vegetation management using disking and mowing to provide suitable waterfowl habitat and promote food sources such as desirable invertebrate communities and seed producing plant species (i.e., annuals; Fredrickson and Reid 1988, Fredrickson 1991).

Disturbance due to moist-soil management affects FQAI, with a high FQAI value is attributed to wetlands supporting native species with low disturbance tolerance (Andreas and Lichvar 1995). Because WRP wetlands were disturbed due to mowing, disking, and manipulation of water levels by water control structures, more disturbance tolerant species (i.e., species with a low coefficient of conservatism) were found in these sites. For example, 46% of plants in WRP wetlands were considered disturbance tolerant (i.e, having a coefficient of conservatism of 0 or 1) compared to 11% of plants in natural wetlands being disturbance tolerant. Similarly, Balcombe et al. (2005b) reported a lower FQAI in mitigation wetlands than in reference wetlands in West Virginia. This difference was attributed to disturbance of the wetlands during their creation and to the young age of mitigation wetlands. The FQAI of mitigation sites was expected to increase with time as the sites recovered from the initial disturbance during their creation

(Balcombe et al. 2005b). Hartzell et al. (2007) reported similar FQAI between older created wetlands (≥ 20 years old) and natural wetlands. The FQAI in WRP wetlands of this study are not likely to increase with time because management practices will continue to cause disturbance on these sites, as evidenced by a lack of correlation between WRP age and FQAI (r = 0.001, P = 0.998).

The reason for higher visual obstruction levels in natural wetlands was the greater proportion of cattails and, to a lesser extent, giant cutgrass in these sites. Many landowners actively managed against cattails growing abundantly in their WRP wetlands to enhance the wetlands for waterfowl, thereby decreasing the mean visual obstruction. The lower proportion of perennial plants and, similarly, higher proportion of annual plants can also be attributed to moist-soil management disturbance. Annual plants are more tolerant of disturbance than perennial species, and annuals are the first species to colonize after disturbance, while perennial species occur later in succession (Holechek et al. 2004). Gray et al. (1999) reported that tilling, disking, and mowing produced more annual grasses in moist-soil wetlands compared to no vegetation management in wetlands.

The proportion of non-native plants was similar between WRP wetlands and natural wetlands. Values recorded in my study were slightly lower than those reported for created (23%) and natural depressional (14%) wetlands in central Oklahoma (Hartzell et al. 2007) and similar to those reported for mitigation (18%) and reference (3%) wetlands in West Virginia (Balcombe et al. 2005b). The similarity of the proportion of non-native plants between wetland types was unexpected because disturbance facilitates establishment and spread of exotic plant species (Hobbs and Humphries 1995). Perhaps non-native species established in both treatments equally because both wetland types were equally distant from the Deep Fork River, a major seed source.

The three most common non-native species in my wetlands were balloon vine, bermudagrass (*Cynodon dactylon* (L.) Pers.), and barnyardgrass (*Echinochloa crus-galli* (L.) P. Beauv.). Balloon vine accounted for 58% of all exotic species observed and was recorded in four natural and eight WRP sites. Bermudagrass composed 22% of all non-native species observed and was recorded in one natural site (1% of all plants in the wetland) and two WRP sites (16% and 15% of all plants in the respective wetlands). One explanation for the presence of bermudagrass is that NRCS used it to vegetate some of the WRP dikes after construction. Also, it could be a remnant of previous land use because bermudagrass is a common forage species in Oklahoma (Tyrl et al. 2008). Barnyardgrass accounted for 9% of all exotic species observed, and was recorded in two natural and four WRP sites. Barnyardgrass is an important food for waterfowl (Mushet et al. 1992), which spread consumed seeds into wetlands via their digestive tracts (Amezaga et al. 2002).

Similarity in plant assemblages between WRP wetlands and natural wetlands was low due to the 52 species that were found solely in WRP wetlands and 19 species that were recorded only in natural wetlands. Similarly, Hartzell et al. (2007) calculated a 38% similarity of plant assemblages between created and natural depressional wetlands in central Oklahoma. The low similarity observed in my study sites is not surprising because of management strategies in WRP wetlands. Management of water levels in WRP wetlands caused different hydroperiods compared to natural wetlands, and direct management of vegetation in WRP wetlands also caused greater disturbance than in natural wetlands. Both management strategies likely caused different plant species to be present in WRP sites than natural sites, as evidenced by the differing plant species richness, FQAI, proportion perennial plants, proportion annual plants, and proportion OBL plants between WRP and natural wetlands.

Interspersion

WRP and natural wetlands did not differ in the percent of wetland area within 3 km of the study site. WRP wetlands are being restored in similar landscapes as natural sites, which demonstrates that WRP wetlands perform the function "maintaining wetland interspersion" at a similar level to natural wetlands. The similar level of interspersion between treatments is, in part, due to them being situated in the Deep Fork River floodplain, which causes a corridor of wetlands maintained by flooding. Also, WRP and natural wetlands were equally distributed along the Deep Fork River and sites tended to cluster together. Buffers of clustered study sites often overlapped. Because of the level of overlap and the equal distribution of treatments, interspersion did not vary between WRP and natural wetlands.

In the Prairie Pothole Region, the function "habitat interspersion and connectivity among wetlands" did not appear to be significantly different between restored WRP and CRP wetlands and reference wetlands (Eckles et al. 2002). Eckles et al. (2002) reported that the restoration of Prairie Pothole Region wetlands within wetland complexes can increase the nesting success of waterfowl due to their need for wetland complexes as opposed to isolated wetlands. Also, wetlands built within larger complexes can also facilitate the movement of biota between wetlands (Amezaga et al. 2002). WRP and natural wetlands in my study were within wetland complexes of similar size, allowing WRP wetlands to provide similar habitat landscapes for waterfowl and other biota as natural wetlands.

CONCLUSIONS

Along the Deep Fork River in Oklahoma, actively managed WRP wetlands are providing the biotic functions of "maintaining characteristic waterbird communities," "maintaining characteristic plant communities," and "maintaining wetland interspersion" at a similar level to natural wetlands. This shows that, in terms of biotic functionality, NRCS is fulfilling its purpose of restoring and enhancing public and private wetlands along the Deep Fork River. WRP wetlands are being restored and enhanced in wetland complexes of similar size as natural wetlands, allowing them to maintain wetland interspersion. Their proximity to other wetlands in the landscape permits organisms to use WRP wetlands as they move through the landscape.

WRP wetlands are also providing beneficial waterbird and plant habitat, allowing them to maintain waterbird and plant communities. Waterbird use of WRP wetlands was similar to natural wetlands, as evidenced by the similar waterbird abundance and richness between wetland types. This also shows that NRCS is meeting its goal of restoring habitat for migratory birds (King et al. 2006). However, some notable differences existed in the composition of waterbirds between wetland types, with WRP wetlands exhibiting a smaller proportion of passerines and a larger proportion of shorebirds than natural wetlands. WRP wetlands are providing habitat for species, such as shorebirds, that is not being provided by natural wetlands in the landscape. Plant species, many of which are wetland dependent, have also become established in WRP wetlands within 3 years after restoration. WRP wetlands exhibited a greater plant species richness and diversity as well as similar evenness to natural wetlands, showing that WRP wetlands are maintaining plant communities. However, some differences in plant community composition existed, including WRP sites exhibiting a lower FQAI, visual obstruction, proportion of perennial plants, and proportion of OBL plants, as well as a higher proportion of annual plants than natural wetlands. This suggests that WRP wetlands have a different composition of plant species, but still provide habitat for a wider range of plant species

I recommend that NRCS continue monitoring WRP wetlands with routine, site-specific assessments. Routine monitoring is an examination of wetland conditions that will allow NRCS and landowners to identify problems before they become an issue (Kentula et al. 1992). For example, invasive species can be identified early and control measures can be set in place before the plants have a chance to spread throughout the entire wetland.

It is important to note that my research focused on actively managed, herbaceous, riverine wetlands along the Deep Fork River, Oklahoma. Because wetlands vary significantly across the country and different factors influence functions for different wetland types, my findings cannot necessarily be applied to other types of WRP wetlands in other regions (Smith et al. 1995). Also, because my study WRP sites were all actively managed, this research cannot comment on the effectiveness of unmanaged WRP wetlands.

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TABLES

Table 1. Characteristics of natural and WRP riverine wetlands located along the Deep Fork River in central Oklahoma, 2009 – 2010.

Treatment	Site No.	County	Approx. Area (ha)	Hydrologic Mgmt.?	Years Since Restoration[a]	No. of Mgmt. Units
Natural	1	Lincoln	17.1	No	--	--
Natural	2	Lincoln	10.8	No	--	--
Natural	3	Lincoln	12.2	No	--	--
Natural	4	Lincoln	8.6	No	--	--
Natural	5	Lincoln	2.9	No	--	--
Natural	6	Creek	1.1	No	--	--
Natural	7	Okfuskee	13.0	No	--	--
Natural	8	Okmulgee	20.2	No	--	--
WRP	1	Lincoln	39.7	Yes	11	2
WRP	2	Lincoln	17.5	Yes	3	2
WRP	3	Lincoln	91.3	Yes	6	3
WRP	4	Lincoln	4.2	Yes	3	3
WRP	5	Lincoln	35.1	Yes	12	2
WRP	6	Lincoln	1.4	Yes	4	1
WRP	7	Creek	27.7	Yes	10	4
WRP	8	Okmulgee	8.5	Yes	8	1

[a] Years since restoration calculated from the start of the first field season.

Table 2. The waterbird species comprising over 10% of those observed overall and during the breeding season, fall migration, and spring migration in natural and WRP wetlands along the Deep Fork River, Oklahoma from May 2009 to September 2010.

Season	Treatment	Common Name	Scientific Name	Proportion of Waterbirds Observed
Overall	Natural	Red-winged blackbird	*Agelaius phoeniceus*	24.8
Overall	Natural	Mallard	*Anas platyrhynchos*	21.6
Overall	Natural	Gadwall	*Anas strepera*	19.7
Overall	WRP	Red-winged blackbird	*Agelaius phoeniceus*	18.6
Overall	WRP	American coot	*Fulica americana*	15.9
Overall	WRP	Mallard	*Anas platyrhynchos*	14.9
Breeding	Natural	Red-winged blackbird	*Agelaius phoeniceus*	37.0
Breeding	Natural	Little blue heron	*Egretta caerulea*	14.1
Breeding	Natural	Wood duck	*Aix sponsa*	11.2
Breeding	WRP	Red-winged blackbird	*Agelaius phoeniceus*	36.1
Breeding	WRP	Snowy egret	*Egretta thula*	12.2
Breeding	WRP	Killdeer	*Charadrius vociferus*	10.4
Fall	Natural	Red-winged blackbird	*Agelaius phoeniceus*	34.0
Fall	Natural	Gadwall	*Anas strepera*	27.6
Fall	Natural	Mallard	*Anas platyrhynchos*	20.2
Fall	WRP	Mallard	*Anas platyrhynchos*	21.1
Fall	WRP	Red-winged blackbird	*Agelaius phoeniceus*	16.1
Fall	WRP	American coot	*Fulica americana*	15.7
Spring	Natural	Mallard	*Anas platyrhynchos*	34.1
Spring	Natural	Gadwall	*Anas strepera*	20.1
Spring	Natural	Blue-winged teal	*Anas discors*	11.6
Spring	WRP	American coot	*Fulica americana*	34.9
Spring	WRP	Blue-winged teal	*Anas discors*	18.8

Table 3. Comparison of waterbird metrics between WRP and natural wetlands along the Deep Fork River, Oklahoma from May 2009 to September 2010. N = 48.

	WRP		Natural			
	Mean	SE	Mean	SE	$F_{1, 47}$	P
Abundance	74.0	18.3	32.2	8.1	1.37	0.249
Richness	5.2	0.7	3.1	0.5	1.05	0.313
Evenness	2.6	0.2	2.2	0.2	0.28	0.599
Diversity	1.0	0.1	0.8	0.1	0.12	0.727

Table 4. Comparison of proportions (%) of waterbird groups between WRP and natural wetlands along the Deep Fork River, Oklahoma from May 2009 to September 2010. An asterisk (*) indicates significance at $P \leq 0.05$. N = 16.

	WRP		Natural			
	Mean	SE	Mean	SE	W	P
Waterfowl	48.3	7.5	35.0	9.9	60.0	0.430
Passerine	17.7	5.6	49.4	11.7	90.0	0.024*
Wading bird	15.3	7.6	10.1	3.8	62.0	0.563
Shorebird	8.6	3.8	1.8	1.1	46.0	0.023*
Other[a]	10.0	4.7	3.7	2.2	62.5	0.590

[a] Other includes American coot, American white pelican, belted kingfisher, double-crested cormorant, gull species, pied-billed grebe, rail species, and sora.

Table 5. The plant species comprising over 5% of those observed in natural and WRP wetlands along the Deep Fork River, Oklahoma from May 2009 to September 2010.

Treatment	Common Name	Scientific Name	Proportion of Plants Recorded
Natural	Southern cattail	*Typha domingensis* Pers.	23.5
Natural	American lotus	*Nelumbo lutea* Willd.	16.8
Natural	Fox sedge	*Carex vulpinoidea* Michx.	14.0
Natural	Swamp smartweed	*Polygonum hydropiperoides* Michx.	11.8
Natural	Giant cutgrass	*Zizaniopsis miliacea* (Michx.) Döll & Asch.	8.4
WRP	Pennsylvania smartweed	*Polygonum pensylvanicum* L.	10.8
WRP	Floating primrose-willow	*Ludwigia peploides* (Kunth) P.H. Raven	9.8
WRP	Southern cattail	*Typha domingensis* Pers.	8.5
WRP	Balloon vine	*Cardiospermum halicacabum* L.	8.1
WRP	Sumpweed	*Iva annua* L.	7.9
WRP	Fox sedge	*Carex vulpinoidea* Michx.	7.7
WRP	Swamp smartweed	*Polygonum hydropiperoides* Michx.	7.6
WRP	Pale spikerush	*Eleocharis macrostachya* Britton	5.7

Table 6. Comparison of plant metrics between WRP and natural wetlands along the Deep Fork River, Oklahoma from May 2009 to September 2010. An asterisk (*) indicates significance at $P \leq 0.05$. N = 32.

	WRP		Natural			
	Mean	SE	Mean	SE	$F_{1, 31}$	P
Richness	23.7	2.2	13.6	1.1	4.27	0.050*
Evenness	7.0	0.8	4.5	0.7	3.01	0.095
Diversity	2.2	0.1	1.7	0.1	4.77	0.039*

Table 7. Comparison of plant proportion metrics (%) between WRP and natural wetlands along the Deep Fork River, Oklahoma from May 2009 to September 2010. An asterisk (*) indicates significance at $P \leq 0.05$. N = 16. Degrees of freedom = 1 for all tests.

	WRP		Natural			
	Mean	SE	Mean	SE	W	P
Proportion non-native	14.2	3.7	7.4	3.3	52.0	0.104
Proportion perennial	74.2	4.6	94.8	1.6	98.0	0.002*
Proportion annual	36.1	6.3	12.3	3.4	41.0	0.005*
Wetland Indicator Status[a]						
OBL	51.5	4.2	81.4	7.5	92.0	0.014*
FACW	23.4	3.8	12.1	5.9	52.0	0.104
FAC	15.9	4.1	6.0	2.7	50.0	0.064
FACU	4.6	2.6	0.2	0.1	57.0	0.205
UPL	0.7	0.4	0.0	0.0	51.5	0.054

[a] Wetland Indicator Status: OBL = obligate, FACW = facultative wetland, FAC = facultative, FACU = facultative upland, UPL = upland.

CHAPTER III

ASSESSING POTENTIAL FUNCTIONAL INDICES FOR INCLUSION IN HYDROGEOMORPHIC FUNCTIONAL MODELS OF RIVERINE WETLANDS IN OKLAHOMA, USA

Abstract: Hydrogeomorphic (HGM) assessment is a rapid assessment method of wetland function using models composed of easily measured indicator variables called functional indices. The objective of this study was to determine if relationships existed between functional indices and a direct measure of function (species richness) for the functions "maintaining characteristic plant communities" and "maintaining characteristic waterbird communities" in natural and restored wetlands. Functional indices assessed for the relationship with plant species richness included soil organic matter, total nitrogen, available phosphorus, electrical conductivity, pH, growing season water table, and percent of the growing season the rooting zone was saturated. In natural wetlands, plant species richness was related to electrical conductivity, pH, and percent of the growing season the rooting zone was saturated. In restored wetlands, plant species richness was related to pH and growing season water table. Waterbird species richness was determined during three seasons (breeding season, fall migration, and spring migration) and assessed for the relationship with plant species richness, vertical structure of the plant

community, floristic quality assessment index, proportion of the wetland that is open, proportion of the plants that are annual, seasonal hydroperiod, seasonal water table, and proportion of the surrounding landscape that is wetland. Waterbird species richness was related to plant species richness, seasonal water table, and seasonal hydroperiod during each season; vertical structure of the plant community during fall migration; and floristic quality assessment index during spring migration. Due to generally weak relationships between plant species richness and functional indices, HGM assessment using the functional indices tested here is not successfully measuring the function "maintaining characteristic plant communities." Due to generally strong relationships between functional indices and waterbird species richness during each season, HGM assessment using the indices I tested is successfully measuring the function "maintaining characteristic waterbird communities" for each season separately.

Key words: functional indices, hydrogeomorphic assessment, Oklahoma, plant species richness, riverine wetlands, waterbird species richness, wetland function

INTRODUCTION

Wetland assessment and monitoring is vital for determining wetland condition and is an important component of wetland management, restoration, and creation (Cronk and Fennessy 2001). Due to the complexity of wetlands and variety of assessment goals, numerous techniques such as indices of biotic integrity (IBI), wetland evaluation technique (WET), and individual state rapid assessment methods have been created to assess wetlands. Assessment methods range from intensive biological and physio-chemical surveys (e.g., IBI) to rapid assessments of wetland function (e.g., WET, hydrogeomorphic [HGM] assessment, and Ohio Rapid Assessment Method; Fennessy et al. 2004).

The HGM approach to assessing wetland functions has four essential components: classification of wetlands based on hydrogeomorphic features, description of functions for wetlands being considered, development of a reference system, and development of an assessment model and functional indices (Brinson 1993). Under HGM, wetlands are first grouped into a class based on geomorphic setting, water source, and hydrodynamics, and then into a subclass based on additional ecosystem and/or landscape characteristics (Smith et al. 1995). Reference wetlands are then determined from within a subclass and include the entire range of wetland conditions (i.e., degraded to pristine conditions; Brinson 1993, Smith et al. 1995). Reference standard wetlands are the least altered wetlands in the least altered landscapes and have the highest level of function across the suite of functions (Smith et al. 1995).

The core of HGM assessment is the functional model, which is used to estimate the functional capacity of a wetland, or how well a wetland performs a particular function (Smith et al. 1995). A functional model is an algorithm composed of easily measured conceptual and/or quantitative variables (functional indices) that contribute to functional capacity (Smith et al. 1995). Functional indices based on structural components are used to predict how well a wetland is performing a particular function because, although direct measures of functional capacity are the most effective technique, they are often too difficult to determine or demand too much time to collect. The range of each functional index is based on reference wetland conditions and has a corresponding value of 0.0 to 1.0, where 1.0 reflects a condition closest to that of reference standard wetlands.

An Assessment Team (A-Team) composed of an interdisciplinary group of scientists creates functional models for wetland subclasses. HGM assessment only occurs within a subclass to control variability and ensure differences observed during assessment are not due to inherent differences between wetlands. The A-Team is responsible for classifying wetlands, identifying reference wetlands, constructing functional models, and calibrating the models within a particular subclass (Smith et al. 1995). Results of the A-Team findings and decisions are published as a guidebook that wetland scientists can use to apply HGM assessment within the specified subclass (Smith et al. 1995).

Validating functional models and their variables should be ongoing during model development (Wakeley and Smith 2001). One approach to model validation is to make certain the indices used in a model do, indeed, relate to the function the model is attempting to measure. Cole (2006) suggested that indices be tested for the relationship with function because even the most basic relationships with function are still untested. Instead, indices are chosen based on the best professional judgment of the A-Team (Cole 2006). Without testing indices, there is no way to know whether a model variable relates to wetland function (Cole 2006). Furthermore, knowing which functional indices are correlated with a function gives developers of models (i.e., the A-Team) a better idea of which indices should be included in the model.

The objective of my study was to determine if potential functional indices that could be used in HGM functional models are related to a direct measure of species richness for the biological functions “maintaining characteristic plant communities” and “maintaining characteristic waterbird communities” for riverine wetlands in central Oklahoma. These two functions represent how well a wetland supports and provides an environment for characteristic communities of plants and waterbirds, respectively (Wilder and Roberts 2002). They were chosen because an appropriate direct measure of function could be calculated. Because an HGM guidebook has not been created for riverine wetlands in central Oklahoma, I selected potential functional indices based on guidebooks for other subclasses and from wetland literature (Casanova and Brock 2000, Colwell and Taft 2000, Gilbert et al. 2006, Hauer et al. 2002, Johnson and Leopold 1994, Kaminski and Prince 1981, Lin 2006, Wilder and Roberts 2002).

METHODS

Study Area

I collected data from 16 herbaceous, riverine wetlands located along the Deep Fork River in central Oklahoma. Eight of the wetlands were restored under Natural Resources Conservation Service’s (NRCS) Wetlands Reserve Program (WRP) and the other eight wetlands were naturally occurring. The sites were located within the Cross Timbers and Cherokee Prairie Major Land Resource Areas (Soil Conservation Service 1979) along the Deep Fork River and spanned 80 km within Lincoln, Creek, Okfuskee, and Okmulgee counties. All sites were flooded at least once every 5 years (Steve Barner, NRCS, personal communication; Bruce Burton, ODWC, personal communication; Darren Unruh, USFWS, personal communication).

Restored wetlands were modified by constructing dikes, excavating depressions, and inserting water control structures. The wetlands were restored 3 to 12 years prior to initiation of this study and the mean age of restored wetlands is 7 years since restoration. Each restored wetland contained between one and four management units, but only one management unit was randomly selected for inclusion in this study. The size of management units ranged from 1 to 20 ha, with a mean size of 8 ha. According to NRCS personnel, all units were actively managed by manipulating water using water control structures and, in some instances, pumping water from the Deep Fork River. Some restored wetlands were also managed by mowing or disking the vegetation during drawdown periods. The size of natural wetlands ranged between 3 and 20 ha, with a mean size of 15 ha. None of the natural wetlands were hydrologically managed.

The plant communities in the wetlands were dominated by herbaceous vegetation and contained only occasional trees or shrubs. The region is characterized by warm, humid summers and mild winters, with a mean annual temperature of 15°C and a mean precipitation of 107 cm (Oklahoma Climatological Survey 2001). The major soil types along the Deep Fork River are Eufaula-Dougherty-Konawa, Osage-Verdigris, and Stephenville-Darnell-Niotaze (Carter and Gregory 2008). All wetland sites are underlain by frequently or occasionally flooded soils (NRCS 2007).

Plant Community

To directly measure the biological function "maintaining characteristic plant communities," I conducted vegetation surveys to determine plant species richness on a larger scale (LSVEG) using the point-step method along transects (Bonham 1989, Smith and Haukos 2002) and on a smaller scale (SSVEG) using plot data. At each wetland, I located two transects that traversed the site perpendicularly to the elevational gradient within the wetland. Along each transect, I recorded the occurrence of a plant species, detritus, bare ground, or open water every one meter (Smith and Haukos 2002). I conducted vegetation surveys in July – October 2009 and in August – September 2010. Plant species richness determined from transect data (i.e., LSVEG) were assessed with functional indices on a large scale because functional indices are meant to relate to the functional capacity of the entire wetland. I also randomly located four circular, 1-m radius plots that were paired with observation wells and soil samples throughout the wetland. Plant species richness of plot data (i.e., SSVEG) were paired with an observation well and soil sample data because indirect indicators such as water table depth and soil saturation may have a more small scale, localized effect on plant species richness (King et al. 2004). Although HGM assessment focuses on functions at a larger scale such as the transect data that were collected, the smaller scale plot data can still provide insight if functional indices are related to local plant communities. Additionally, SSVEG was evaluated in the event that the sample size was too small to observe relationships at a larger scale. Points were randomly selected using Minnesota Department of Natural Resources (DNR) Random Sample Generator version 2.2 (Minnesota DNR, St. Paul, Minnesota, USA) in ArcView version 3.3 (ESRI, Redlands, California, USA). At each plot, I recorded all plant species and estimated their percent cover. I conducted vegetation surveys of the plots from August – September 2009. Plant species richness at the small and large scales was calculated as the number of species observed in each plot and wetland, respectively. Species richness was $\log_{10}$-transformed for both metrics (Matthews et al. 2005).

Waterbird Community

To directly measure the biological function "maintaining characteristic waterbird communities," I conducted waterbird surveys to determine waterbird species richness during the breeding season (BSBIRD), fall migration (FMBIRD), and spring migration (SMBIRD). In each wetland, I used point counts with unlimited sight distance and no overlap to survey waterbirds (Hartzell et al. 2007). Points were situated to allow for maximum visibility (e.g., on higher ground or in open areas), and the number of points in a wetland ranged from two to four, depending on the size of the wetland and visibility. At each point, all waterbirds seen or heard were recorded during a 10 minute period. During the breeding season, playback calls were used at each point to determine presence of secretive species such as rails and bitterns (Ribic et al. 1999). I walked transects between points to flush hidden birds when vegetation was too dense to observe birds. Species observed along transects were also recorded and were included in species richness numbers. I did not include birds flying over the wetland in the survey data. Surveys were not conducted when there was precipitation or when winds exceeded 25 km/hr (U.S. Environmental Protection Agency 2002). A total of two, four, and two surveys were completed during the breeding season (June – early July 2009), fall migration (mid-August – early December 2009), and spring migration (March – early May 2010), respectively. Breeding season surveys were completed between sunrise and 4 hr after sunrise (Ribic et al. 1999), while fall and spring migration surveys were completed during daylight hours. Surveys were conducted a minimum of 10 days apart (Desrochers et al. 2008).

Waterbird species richness was calculated as the number of species observed per wetland and was averaged over the season. Species richness for each season was $\log_{10}$-transformed or, if no species were present (i.e., species richness = 0) for any sampling period, one was added to all species richness measurements for that period so that the $\log_{10}$ could be calculated, and then $\log_{10}$-transformed (Matthews et al. 2005).

Functional Indices

I analyzed the following functional indices to determine if they exhibited relationships with LSVEG and SSVEG: (1) soil organic matter (SOM), (2) total nitrogen (TN), (3) available phosphorus (P), (4) pH, (5) electrical conductivity (EC), (6) median depth of the water table during the growing season (WT), and (7) percent of time the rooting zone was saturated during the growing season (SAT; Table 1). Soil property measures included SOM, TN, P, pH, and EC because they can impact plant communities and have all been shown to affect plant species richness (Johnson and Leopold 1994, Houlahan et al. 2006, Sutton-Grier et al. 2009). Hydrologic effects were quantified by WT and SAT because the hydrology of the wetland can influence plant species richness (van der Valk et al. 1994, Casanova and Brock 2000). HGM guidebooks have included functional indices of soil properties (e.g., Wilder and Roberts 2002) and hydrologic variables (e.g., Hauer et al. 2002, Wilder and Roberts 2002, Lin 2006) for assessing the function "maintaining characteristic plant communities."

I analyzed the following functional indices to determine if they exhibited relationships with BSBIRD, FMBIRD, and SMBIRD: (1) LSVEG, (2) floristic quality assessment index (FQAI), (3) percent of the plants that are annual (ANNUAL), (4) vertical structure of the plant community (VS), (5) percent of the wetland that is open water or bare ground (OPEN), (6) hydroperiod for each respective season (HP), (7) median height of the water table during each respective season (WT), and (8) percent of the surrounding landscape that is wetland (SURRWL). The indices LSVEG, FQAI, and ANNUAL were meant to reflect the composition of the plant communities because vegetation can control food web dynamics (Gilbert et al. 2006) and plants can provide food for waterbirds (Thomas 1982). The structure of the vegetation was quantified using VS and OPEN, which can impact the waterbird species present in a wetland (Kaminski and Prince 1981, Traut and Hostetler 2003). Hydrologic effects on waterbird communities were measured by HP and WT, which have been shown to affect waterbird species richness (Colwell and Taft 2000, Webb et al. 2010). The surrounding land use was measured by SURRWL because the area of wetland habitat in the surrounding landscape has been shown to affect waterbird species richness (Fairbairn and Dinsmore 2001). HGM guidebooks have included functional indices of plant community composition (e.g., Gilbert et al. 2006), vegetation structure (e.g., Hauer et al. 2002, Lin 2006), hydrologic variables (e.g., Hauer et al. 2002), and surrounding wetlands (e.g., Lin 2006) to assess functions of wildlife use and habitat in wetlands.

Hydrologic Indices. Hydrological functional indices (i.e., WT, SAT, and HP) were determined from monitoring observation wells. I installed observation wells in each of the four randomly selected points used for vegetation plots and monitored each well once per month for one year (June 2009 – May 2010), except in June 2009 when I monitored the wells twice during the month. Wells were constructed following the method outlined by Vepraskas (2005). The wells were constructed from 5-cm polyvinyl chloride (PVC) pipe, slotted every 1.27 cm to 15 cm below the soil surface, covered in mesh screen, and capped on the top and bottom. An auger was used to create an 8.5 cm hole and wells were placed to a depth of 1 m. Coarse sand was placed around the well, bentonite pellets were used to seal the top of the sand, and soil was mounded around the soil surface. During observations, I used a measuring tape to determine the depth of water in the well or depth above the soil surface. I set 0 cm as the soil surface, making all observations below the surface negative and all observations above the surface positive. When the water table was below the bottom of the well, I recorded the water table depth as -105 cm.

At each point, WT was calculated as the median value of WT observations. For relating WT to LSVEG and SSVEG, only observations reported during the growing season were included in the calculation of WT, and for relating WT to BSBIRD, FMBIRD, and SMBIRD, only observations recorded during each respective season were included in calculations of WT. The median was calculated to determine WT because actual depths were not recorded when water was below the well or at a depth greater than 100 cm (i.e., < 100 cm or > 100 cm; Cole and Brooks 2000). I calculated SAT at each point as the percent of the time water was observed at or above –30 cm during the growing season (i.e., standing water or a water table within the top 30 cm of the soil; Cole and Brooks 2000). The top 30 cm of the soil was chosen because this is the major rooting zone (Cole and Brooks 2000). HP was determined at each point as the percent of the time water was greater than 0 cm during each of season.

Soil Indices. Functional indices of soil properties, including SOM, TN, P, pH, and EC, were determined from soil samples taken within the rooting zone at each of the four points used for vegetation plots. For determination of SOM and TN, I collected a soil sample from depths of 5 cm and 20 cm twice during the year (Magee et al. 1993). Soil samples used for determination of P, pH, and EC were collected to a depth of 20 cm (Bruland and Richardson 2006). Soil samples for EC and pH were collected three times throughout the year, while soil samples for P samples were collected twice. All samples were analyzed by Oklahoma State University's Soil, Water and Forage Analytical Laboratory following standard methods. Total organic carbon was determined using the dry combustion method (Nelson and Sommers 1996) and converted to SOM by multiplying by 1.72 (Gosselink et al. 1984); TN was determined using the Kjeldahl Method (Bremner and Mulvaney 1982); P was determined using a Mehlich 3 extract measured on a Spectro ICP; pH was determined using a 1:1 soil-water extract and pH meter; and EC was determined using a saturated soil paste (Gavlak et al. 2003). I calculated SOM, TN, P, pH, and EC at each point as the mean of the observations.

Vegetation Indices. I assessed the functional index for VS using a Robel pole (Robel et al. 1970). The Robel pole was constructed from a 2 m PVC pipe, marked every 5 cm. I recorded visual obstruction measurements by standing at a point 4 m from the pole and observed the pole at a 1 m height from the four cardinal directions (Robel et al. 1970). Data were recorded at six randomly selected points, four of which were the vegetation plot sites. I placed the Robel pole in areas where vegetation was not trampled from my previous visitations of the site. Measurements were conducted September – early October 2009. At each point, VS was determined by taking the mean of the readings from the four cardinal directions. The mean for each wetland was then calculated from all points.

I determined the functional index for FQAI using the methods outlined by Andreas and Lichvar (1995). Each species encountered along vegetation transects was assigned a coefficient of conservatism based on the species' nativity and disturbance tolerance. Coefficients range from 0 to 10, with 0 indicating a non-native species or a native species that becomes an opportunistic invader, and 10 indicating a species with high fidelity and small ecological range (Andreas and Lichvar 1995). Coefficients were based on previously published literature (e.g., Andreas and Lichvar 1995, Hartzell et al. 2007). Coefficients were then combined for each wetland using the FQAI equation described in Andreas and Lichvar (1995). Again, the mean was taken of the four values of FQAI in a wetland to obtain one value for each wetland.

Plant transect surveys were used to determine OPEN and ANNUAL. All points recorded as open water or bare ground were combined to determine the total number of open area points. OPEN was calculated as the total number of open points divided by the total number of points. The percent of plants that were annual was calculated for ANNUAL. The U.S. Department of Agriculture's PLANTS Database was used to determine if plants were annual or perennial (U.S. Department of Agriculture 2011).

Landscape Index. My landscape functional index was the percent of the landscape within 3 km of a study site that was wetland (i.e., SURRWL) because past research has shown wetland area within 3 km of a wetland to be a significant predictor for bird species richness (Fairbairn and Dinsmore 2001). SURRWL was determined in ArcMap version 10 (ESRI, Redlands, California, USA). To determine the position of wetlands in the surrounding area for SURRWL, I used National Wetlands Inventory (NWI) maps (Fairbairn and Dinsmore 2001, Cunningham et al. 2007). Open water observed on digital ortho imagery layers from the National Agriculture Imagery Program (Farm Service Agency, Aerial Photography Field Office, Salt Lake City, Utah, USA), but not included in the NWI, were traced and then added to the NWI coverage (Fairbairn and Dinsmore 2001). Each study wetland was then buffered at 3 km. The percent of the area within the 3 km buffer that was wetland was used to calculate SURRWL.

Wetland Area

Wetland area (AREA) was used in the analysis because area is known to affect plant species richness (Matthews et al. 2005, Houlahan et al. 2006) and waterbird species richness (Webb et al. 2010). AREA was not viewed as a functional index, but simply as a way to help explain variability between wetlands. Wetland area was determined by tracing the perimeter of each site on a digital ortho imagery layer. Species richness measurements and AREA were $\log_{10}$-transformed in order for the relationship between species richness and area to be linear (Matthews et al. 2005).

Data Analyses

Plant communities of restored wetlands were directly affected by management practices, such as mowing, while natural wetlands were not managed. Because the direct management could affect how plant communities respond to functional indices, the plant communities in restored and natural wetlands were analyzed separately. For example, mowing vegetation affects plant species richness, but it was not measured as a functional index. Because mowing may affect plant species richness, the relationship between other functional indices and species richness may change. For example, pH may have a relationship with plant species richness in natural wetlands but that relationship may not occur in restored wetlands because the effects of mowing are outweighing the effects of pH. Management strategies for waterbirds consisted of directly manipulating the water levels and plant communities, causing differing water levels, hydroperiods, amounts of open area, plant communities, and vertical structure of the plant communities between restored and natural wetlands. However, all of these differences have been quantified by the functional indices of WT, HP, OPEN, LSVEG, FQAI, and VS. Because the differences caused by management have been measured, restored and natural wetlands were combined in order to increase the range of conditions for each functional index in analysis.

Prior to analysis, data were plotted against species richness to determine if the relationship was linear or nonlinear to ensure all data exhibited a linear relationship with species richness and met the assumptions of multiple linear regression. An Anderson-Darling test was then used to test for normality of all data (Minitab, Inc. 2010). In natural wetlands, EC calculated at the small scale was rank transformed due to its nonlinear relationship with SSVEG in natural wetlands. I conducted correlations using Pearson's correlation and Spearman's rank correlation to determine if a direct relationship between species richness and each functional index existed. Spearman's rank correlation was used for non-parametric data and Pearson's correlation was used for parametric data. Alpha values were set at 0.10 in order to reduce Type I error. Relationships were considered strong if r or ρ was ≥ 0.5 (Fairbairn and Dinsmore 2001). Correlations were also conducted between potential indices to reduce multicollinearity in multiple linear regression. Due to its strong correlation with TN, SOM was not used in regression analysis ($\rho = 0.978$, $P < 0.001$). ANNUAL was also not used in regression analysis due to its strong correlation with OPEN ($\rho = 0.991$, $P < 0.001$).

I used multiple linear regression to determine relationships between species richness and functional indices when all indices were present in analysis. LSVEG was not included in multiple linear regression analysis because it had too many predictor variables for the sample size (Minitab, Inc. 2010). Multiple linear regression used the ordinary least squares method that derives the equation by minimizing the sum of the squared residuals (Minitab, Inc. 2010). To determine the best regression model for each of the metrics (SSVEG, BSBIRD, FMBIRD, and SMBIRD), I used the best subsets regression tool to determine the 5 multiple linear regression models with the highest R^2

for each possible number of predictors (e.g., 5 models using 1 variable, 5 models using 2 variables; Matthews et al. 2005). The response variable for the best subsets regression was SSVEG, BSBIRD, FMBIRD, or SMBIRD. AREA was set as a predictor in all models and functional indices were free predictor variables in best subsets regression. I then conducted a multiple linear regression for each of the best fit models. Residuals vs. fits plots and histograms were used to determine residual equal variance and normality, respectively (Minitab, Inc. 2010). Variance Inflation Factors were used to determine multicollinearity (Minitab, Inc. 2010). The model with a p-value ≤ 0.1, all variables with p-values ≤ 0.1, and the highest R^2 was considered the best model for predicting species richness (Matthews et al. 2005).

RESULTS

Plant Community

A total of 70 plant species were recorded along transects and in vegetation plots in natural wetlands. Based on vegetation plot data, a mean of 12 species per natural wetland was calculated and transect data had a mean of 20 species per natural wetland. The most common species in natural wetlands were southern cattail (*Typha domingensis*), American lotus (*Nelumbo lutea*), and fox sedge (*Carex vulpinoidea*), which composed 54% of individuals observed along transects. A total of 83 species were observed along transects and in vegetation plots in restored wetlands. Vegetation plot data averaged 12 species per restored wetland and transect data averaged 16 species per restored wetland.

The most common species in restored wetlands were Pennsylvania smartweed (*Polygonum pensylvanicum*), floating primrose-willow (*Ludwigia peploides*), and fox sedge, which composed 48% of individuals observed along transects.

The only functional index that was correlated with LSVEG in natural wetlands was pH, which exhibited a strong negative correlation (Table 2; see Appendix 3 for scatterplots). No functional indices were correlated with LSVEG in restored wetlands. There was a weak positive correlation between SSVEG and EC and a weak negative correlation between SSVEG and SAT in natural wetlands (Table 3). In restored wetlands, SSVEG exhibited a weak positive correlation with pH and a weak negative correlation with WT. The best multiple linear regression equation for SSVEG in restored wetlands included only WT (R^2 = 0.453; Table 4). None of the best subset models were significant for SSVEG in natural wetlands. For a summary of functional indices used in plant species richness analysis, see Appendix 2.

Waterbird Community

I observed a total of 41 waterbird species in the study wetlands. During the breeding season, 18 species were observed, with sites having a mean of 4 species. The most common species during the breeding season were red-winged blackbird (*Agelaius phoeniceus*), great blue heron (*Ardea herodias*), and common yellowthroat (*Geothlypis trichas*), which composed 41% of individuals observed. During the fall migration, 31 species were observed, averaging 2 species per wetland. The most common species during fall migration were red-winged blackbird, mallard (*Anas platyrhynchos*), and great blue heron, which composed 33% of individuals observed. During the spring migration, I observed 26 species, with wetlands averaging 3 species. The most common species during spring migration were blue-winged teal (*Anas discors*), Canada goose (*Branta canadensis*), red-winged blackbird, and American coot (*Fulica Americana*), which composed 41% of individuals observed.

During the breeding season, LSVEG, HP, and WT were all strongly positively correlated with BSBIRD (Table 5; see Appendix 3 for scatterplots). Multiple linear regression analysis showed that LSVEG and HP explained most of the variation in BSBIRD (R^2 = 0.782; Table 4). During fall migration, LSVEG and HP were both strongly positively correlated with FMBIRD and WT was weakly positively correlated with FMBIRD (Table 6). The two variables in the final model produced by multiple linear regression analysis were VS and WT (R^2 = 0.813). During spring migration, only LSVEG was strongly positively correlated with SMBIRD (Table 7). The best model from multiple linear regression analysis for SMBIRD included LSVEG, FQAI, HP, and WT (R^2 = 0.748). For a summary of functional indices used in waterbird species richness analysis, see Appendix 2.

DISCUSSION

Plant Community

pH. Few relationships existed between plant species richness and potential indices. Plant species richness exhibited a strong negative correlation with pH in natural wetlands at the larger scale and a weak positive relationship in restored wetlands at the smaller scale. Plant species richness has been found to decrease with increasing pH both at lower pH ranges (4.5 – 6.7; Woodcock et al. 2005) and higher pH ranges (6.5 – 7.9; Ashworth et al. 2006). The range of pH values in natural wetlands in my study (5.1 – 8.0) fell between these two and exhibited the same trend. Lower pH levels are characterized by higher concentrations of essential plant micronutrients than more alkaline soils (Montgomery et al. 2001), which is likely the cause for a greater species richness at lower pH levels.

Disturbance can alter the relationship between pH and macrophyte species richness (Woodcock et al. 2005), which may explain why the same trend was not observed in restored wetlands that are subject to water level fluctuations, disking, and mowing. Woodcock et al. (2005) reported that disturbances caused by beavers likely cause the coexistence of species without allowing time for any species to dominate at any particular pH. In my restored wetlands, pH was weakly positively correlated with SSVEG. However, the correlation was largely based on five points, four of which were from one wetland. Based on personal observation, that wetland was the only one to be disked at the beginning of the growing season when I conducted small scale plant surveys. The large plant species richness at this site was likely due to disking and not to pH. When this wetland is removed from the analysis, no correlation existed between pH and SSVEG in restored wetlands ($P = 0.112$). The weak positive correlation between SSVEG and pH was likely due more to the disking of one wetland than to an actual relationship with pH. Likely, pH did not exhibit a true relationship with plant species richness in these sites and should not be included as a functional index in HGM assessment of restored wetlands.

A few differences existed in plants species composition between wetlands with higher mean pH (> 7.7) and lower mean pH (≤ 6.3). Swamp smartweed (*Polygonum hydropiperoides* Michx.), delta arrowhead (*Sagittaria platyphylla* (Engelm.) J.G. Sm.), and common buttonbush (*Cephalanthus occidentalis* L.) were encountered more in wetlands with lower pH than those with higher pH. The only species observed more in sites with higher pH than lower was lanceleaf frogfruit (*Phyla lanceolata* (Michx.) Greene).

WT and SAT. Hydrologic variables had an effect on plant species richness. A weak negative relationship existed between WT and SSVEG in restored wetlands. This relationship is largely caused by very low species richness at high standing water depths. Water depths were recorded at greater than 60 cm at three points, all of which had a species richness of 0. Also, of the three points between 40 cm and 60 cm, species richness ranged from 1 to 3. The findings of this study coincide with others that have shown plant species richness to be higher in shallow wetlands than deeper wetlands (van der Valk et al. 1994, Ashworth et al. 2006). The lack of relationship between WT and SSVEG in natural wetlands or LSVEG in both natural and restored wetlands was likely due to the lower maximum WT, which never exceeded 35 cm. Moser et al. (2007) also reported no correlation between water table depths and plant species richness when maximum water depth was less than 30 cm. Similarly, SAT exhibited a weak negative relationship with SSVEG in natural wetlands. Plant species richness was shown to decrease as the duration of flooding during the growing season increased from 0% to 100% of the growing season (Casanova and Brock 2000). Longer flooding events allow only water tolerant species to germinate, while shorter flooding events allow time for terrestrial seeds to also germinate (Casanova and Brock 2000). In my study, SAT ranged from 20% to 100% of the growing season and exhibited a significant positive correlation with the percent of obligate wetland plants ($\rho = 0.667$, $P < 0.001$), demonstrating that longer hydroperiods are only allowing obligate plants to grow, thereby decreasing species richness.

EC. In natural wetlands, SSVEG was weakly positively correlated with EC. Johnson and Leopold (1994) reported a "hump" shaped curve when relating EC to plant species richness in natural wetlands, with a peak in species richness around 4.1 dS/m. My study exhibited the same trend prior to rank transformation with the highest species richness occurring when EC was approximately 4.5 dS/m. Johnson and Leopold (1994) suggest the reason for the curvilinear relationship between EC and plant species richness was based on Grime's (1973) model, later coined the "intermediate disturbance hypothesis," which states that intermediate ranges of stress will result in the highest species richness because low levels of stress result in more competitive species forming monocultures and high levels of stress will result in only a few species that can survive such conditions. In my study wetlands, the main monoculture forming species was southern cattail, which occurred when EC was relatively low (1.1 – 2.1 dS/m). Additionally, high levels of EC were likely affecting which plant species were present in my study wetlands. For example, salt heliotrope (*Heliotropium curassavicum*), which has a high salinity tolerance (U.S. Department of Agriculture 2011), was only found in wetlands with a mean EC of > 7.0 dS/m, and saltcedar (*Tamarix ramosissima* Ledeb.), which is also salt tolerant (Tyrl et al. 2008), was present in a wetland with an EC of 11.0 dS/m. Monoculture species occurring at lower levels of EC and the presence of salt tolerant species at higher levels of EC suggest that Grime's model may explain the effect of EC on plant species in my study wetlands. I should note that the intermediate range of EC was not represented in my wetlands, with a gap in EC from 3.0 – 6.0 dS/m, which may have impacted the results by changing the curve of the relationship.

Another possible explanation for the relationship between EC and plant species richness is the interaction of EC and SAT. EC exhibited a weak, negative relationship with SAT ($\rho = -0.397$, $P = 0.027$) as well as with the proportion of the entire year the rooting zone was saturated ($\rho = -0.341$, $P = 0.061$). This relationship was likely because salts dissolve in water and, because of that, they were suspended in the water column and not concentrated in the soil. Because SAT also influences plant species richness, the relationship between EC and richness may be partially due to EC and SAT interactions. For HGM assessment, an appropriate way to assess EC in natural wetlands would be to assign a value to the functional index based on the "humped" curve with 1.0 being assigned when EC is near 4.5 dS/m and decreasing as EC moves from 4.5 dS/m. Plant species richness only exhibited a nonlinear relationship with EC analyzed at the small scale in natural wetlands.

In restored wetlands, EC was not related to LSVEG or SSVEG. Similarly, Ashworth et al. (2006) found EC was not significantly contributing to plant species richness in created wetlands. Perhaps the reason for the lack of relationship between EC and plant species richness in restored wetlands in my study was due to the effects of management in restored sites. The final regression model only included WT and AREA in the final model for SSVEG in restored wetlands, implying that management of the WT was outweighing the effects of EC. EC was only weakly correlated with SSVEG in natural wetlands so it would stand to reason that other factors could outweigh the effects of EC on plant species richness.

Waterbird Community

LSVEG. All three vegetation indices were related to waterbirds during at least one season. LSVEG was strongly related to waterbird species richness during all seasons. LSVEG was likely related to waterbird species richness because it was strongly, positively correlated with a visual estimation of the number of vertical vegetation zones in each wetland (deep water, shallow water or submergent plants, short emergent plants, medium emergent plants, tall emergent plants, and trees or shrubs; $r = 0.767$, $P = 0.001$). The larger number of zones in wetlands with higher LSVEG allowed a wider range of waterbird species to inhabit those sites (Weller 1999). Perhaps a more applicable and rapid functional index for HGM assessment would be to visually estimate the number of vegetation zones instead of using LSVEG.

VS. The relationship of VS with waterbird species richness varied by season, which was determined to have a negative relationship during fall migration and no relationship during the breeding season and spring migration. The difference between seasons is due to changes in species' preferences and species composition between seasons. Some species, such as red-winged blackbird, common yellowthroat, and green herons

(*Butorides virescens)*, prefer tall emergent vegetation such as cattails (Traut and Hostetler 2003, Maddox and Wiedenmann 2005, Safratowich et al. 2008). Other species, such as wading birds other than green heron, have been reported to prefer open water over tall emergent vegetation (Traut and Hostetler 2003). Dabbling duck preferences change throughout the year, preferring greater cover during the breeding season and areas of open water during the autumn (Ross and Merkin 2009). The negative relationship between FMBIRD and VS can be explained by the high proportion of waterbird species that prefer open water occupying study wetlands during fall migration. The lack of relationship of VS with BSBIRD and SMBIRD is explained by the presence of both species preferring tall emergent vegetation and species preferring open water during those seasons. It is important to note that all wetlands with high average VS had at least some open water in which species preferring sparser vegetation could inhabit.

FQAI. The final regression model for SMBIRD showed a positive influence of FQAI on SMBIRD. FQAI is considered a reflection of a site's level of anthropogenic disturbance (Matthews et al. 2005). Low FQAI values indicate higher disturbance than high FQAI values, demonstrating that SMBIRD is being negatively influenced by anthropogenic disturbance because as FQAI decreases (i.e., disturbance increases), SMBIRD decreases. Hartzell (2006) also found that a negative relationship between disturbance and avian species richness that varied by season. The reasons Hartzell (2006) cited for varying relationships between seasons were different bird communities using the wetlands during each season and varying hydroperiods throughout the year, which was not accounted for in analysis. In my study, however, the variable hydroperiod was already accounted for in multiple linear regression. The differing relationships between FQAI and waterbird species richness between seasons was likely due to differences in species composition and species' preferences throughout the year.

WT and HP. Waterbird richness was positively influenced by WT during all seasons and HP during the breeding season and fall migration, but was negatively influenced by HP during spring migration. The positive influence of seasonal WT on each season's waterbird species richness is because water levels were measured both below and above the soil surface, so higher numbers indicate standing water, which is important for many waterbird species. Other studies have reported a negative relationship between average water depth and waterbird species richness in emergent wetlands (Colwell and Taft 2000, Webb et al. 2010). However, the average water depth for each of the wetlands measured in those studies was above the soil surface. Low water levels in my study indicated no standing water, while low water levels in Colwell and Taft (2000) and Webb et al. (2010) indicated shallow water. When WT and HP are combined, the relationship more closely matches that of Colwell and Taft (2000) and Webb et al. (2010). During the breeding season and fall migration, WT values were relatively low with a median WT of -46 cm and 3 cm, respectively, and maximum of 17 cm and 42 cm, respectively. During these seasons, HP was positively related to waterbird richness. However, during spring migration WT values were relatively higher, with a median of 29 cm and maximum of 96 cm. During spring migration, HP was negatively related to waterbird richness. When WT and HP are viewed together, they indicate that shallow standing water is likely preferable for waterbird species. For HGM assessment, functional models should combine WT and HP to properly assess the effects of hydrology on waterbird communities.

SURRWL. Multiple linear regression and correlation analyses did not suggest waterbird species richness was related to SURRWL during any season. The importance of the amount of wetlands within 3 km of a site on waterbird species richness varies by area, with some areas demonstrating a positive relationship (Fairbairn and Dinsmore 2001) and others exhibiting no relationship (O'Neal et al. 2008). Perhaps if the width of the buffer around each wetland used in SURRWL analysis was changed, a relationship may emerge. For example, bird species richness has been positively related to percent of wetland area within 1 km (Mensing et al. 1998) and 10 km (Webb et al. 2010) of a wetland.

CONCLUSION

In conducting this study, I attempted to use the most robust sampling method based on the preferred methods outlined in HGM assessment. An assumption I made was that the more comprehensive a measurement of functional index was, the more likely it would directly relate to species richness. For example, for the measurement of water table depth, Wilder and Roberts (2002) first prefer using groundwater monitoring well data and, if that is not possible, to determine the depth at which abundant redoximorphic features occur. However, as Wilder and Roberts (2002) point out, redoximorphic features may be reflecting past water tables and not current conditions. I chose the preferred measurement of using monitoring wells to determine water table depth to increase accuracy. Because I only used the more robust method to measure functional indices, the relationship between more rapid measurement methods of functional indices and function is still unknown. Future research will need to determine if more rapid assessment methods relate to biotic wetland functions in order to determine their applicability to HGM assessment.

Plant species richness exhibited weak to no relationships with most of the functional indices used in HGM assessment that were tested in this study. One possible explanation for weak relationships was that important variables were missing from analysis (Kaźmierczak et al. 1995). For example, Casanova and Brock (2000) found that short frequent floods caused higher plant species richness than long infrequent floods. Perhaps if flooding frequency was included as a functional index in multiple linear regression, a stronger relationship between plant species richness and hydrological variables would emerge. However, based on the fact that most of my variables exhibited such weak relationships with plant species richness, just a few more variables will likely not affect my results to a large extent. Too many variables are affecting plant communities to be able to measure the relationship between wetland structural variables and plant communities in a short enough time period to be effective in HGM assessment. Given the weak relationships between functional indices and plant species richness, HGM assessment using the functional indices I tested is not successfully measuring the function “maintaining characteristic plant communities.”

Waterbird species richness was related to most of the functional indices, demonstrating that these indices are appropriate for use in HGM assessment. My present multiple linear regression models explained 75% – 81% of the variability between wetlands. The functional indices in the final regression models are appropriate for quantifying the function "maintaining characteristic waterbird communities" in HGM assessment. However, these relationships varied by season due to changing composition and species' requirements. Because of the changing wetland habitat requirements for waterbirds within each season, one functional model is not applicable to all seasons. Perhaps a more appropriate approach is to create a functional model for each season using the functional indices included in the final multiple linear regression model. If a wintering waterbird HGM model is desired, future research should also assess the relationship of functional indices on wintering waterbird communities to determine the applicability of indices for that season.

Currently, all HGM guidebooks listed on the U.S. Army Corps of Engineer's website (http://el.erdc.usace.army.mil/wetlands/guidebooks.cfm) include all wildlife in one function such as "maintaining characteristic fauna." I focused only on waterbirds because directly assessing all wildlife communities was outside the scope of this study. Based on the fact that relationships between functional indices and waterbird species richness varied by season and that species' requirements vary so much between assemblages, it is unlikely that one HGM functional model could be used to capture the variability between all wildlife during all seasons. For example, a functional index for amphibians could be the percent of the surrounding area that is urbanized, which was found to negatively affect amphibian species richness (Mensing et al. 1998). However, the same metric of urbanization has been found to positively affect fish species richness (Mensing et al. 1998). If the amount of urbanization around a wetland was to be a functional index for "maintaining characteristic fauna," there would be no way for the model to reflect urbanization's effect on both fish and amphibians. The function "maintaining characteristic fauna" must be narrowed down into more specific functions for each wildlife group in order for HGM assessment to properly assess wildlife functions in wetlands.

Limitations do exist in this study that should be discussed. First, species richness was used as a direct measure of function, but this is not a comprehensive metric. For example, it does not take into account evenness, diversity, abundance, presence of threatened or endangered species, or presence of invasive species. However, no better single measure could be determined. Another limitation is that no reference or reference standard wetlands have been identified for the region. Because of this, the full range of conditions may not be represented. Also, alpha values were set at 0.1 in order to decrease Type I error, which invariably means Type II error increased. If the A-Team for this region chooses to further refine relationships between function and functional indices to reduce Type II error, they can now focus on those already shown to exhibit relationships based on my study.

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TABLES

Table 1. The HGM functional indices and their abbreviations included in correlation analysis for relationship with plant species richness analyzed at two scales and waterbird species richness analyzed during three seasons in riverine wetlands along the Deep Fork River, Oklahoma, USA.

Direct measures and potential indices	Abbreviation
Plant species richness, log_{10} transformed vs.:	
Soil organic matter (%)	SOM
Total nitrogen (%)	TN
Available phosphorus (ppm)	P
pH	pH
Electrical conductivity (dS/m)	EC
Median height of the water table during the growing season (cm)	WT
Percent time rooting zone is saturated during the growing season (%)	SAT
Waterbird species richness, log_{10} transformed vs.:	
Large scale plant species richness	LSVEG
Vertical structure of the plant community (dm)	VS
Floristic quality assessment index	FQAI
% of the wetland that is open water or bare ground (%)	OPEN
% of the plants that are annual (%)	ANNUAL
Hydroperiod during the season (%)	HP
Median height of the water table during the season (cm)	WT
% of the surrounding 3 km that is wetland (%)	SURRWL

Table 2. Spearman's rho or Pearson's *r* and P-values for large scale vegetation richness (LSVEG) correlation with functional indices in riverine wetlands along the Deep Fork River, Oklahoma, USA (n = 8). SOM = soil organic matter, TN = total nitrogen, P = available phosphorus, EC = electrical conductivity, WT = median growing season water table, SAT = percent of the growing season the rooting zone was saturated. * indicates significance at $P \leq 0.10$. † indicates a strong relationship at $\rho \geq 0.50$ or $r \geq 0.50$.

	ρ or *r*[a]	P
SOM		
Natural wetlands	0.313	0.450
Restored wetlands	-0.500	0.207
TN		
Natural wetlands	0.313	0.450
Restored wetlands	-0.524	0.183
P		
Natural wetlands	-0.578	0.133
Restored wetlands	0.080	0.851
EC		
Natural wetlands	-0.602	0.114
Restored wetlands	0.024	0.955
pH		
Natural wetlands	-0.651†	0.081*
Restored wetlands	-0.048	0.911
WT		
Natural wetlands	0.169	0.690
Restored wetlands	0.095	0.823
SAT		
Natural wetlands	0.358	0.384
Restored wetlands	0.000	1.000

[a] P correlations were analyzed using Pearson correlation. All other data (SOM, TN, EC, pH, SAT, and WT) correlations were analyzed using Spearman correlation.

Table 3. Spearman's rho or Pearson's r and P-values for small scale vegetation richness (SSVEG) correlation with functional indices in riverine wetlands along the Deep Fork River, Oklahoma, USA (n = 32). SOM = soil organic matter, TN = total nitrogen, P = available phosphorus, EC = electrical conductivity, WT = median growing season water table, SAT = percent of the growing season the rooting zone was saturated. * indicates significance at $P \leq 0.10$. † indicates a strong relationship at $\rho \geq 0.50$ or $r \geq 0.50$.

	ρ or r[a]	P
SOM		
Natural wetlands	-0.048	0.799
Restored wetlands	-0.044	0.810
TN		
Natural wetlands	-0.005	0.981
Restored wetlands	-0.157	0.391
P		
Natural wetlands	-0.041	0.825
Restored wetlands	0.150	0.412
EC		
Natural wetlands	0.412	0.021*
Restored wetlands	-0.212	0.245
pH		
Natural wetlands	0.200	0.281
Restored wetlands	0.415	0.018*
WT		
Natural wetlands	-0.161	0.387
Restored wetlands	-0.314	0.080*
SAT		
Natural wetlands	-0.322	0.077*
Restored wetlands	0.058	0.751

[a] P correlations were analyzed using Pearson correlation. All other data (SOM, TN, EC, pH, SAT, and WT) correlations were analyzed using Spearman correlation.

Table 4. Best models determined during multiple linear regression analysis, coefficient of determination (R^2), adjusted R^2, and p-values for predicting small scale plant species richness in restored wetlands as well as breeding season, fall migration, and spring migration waterbird species richness in riverine wetlands along the Deep Fork River, OK, USA. See Table 1 for abbreviations.

	Intercept +/- regression coefficient (Variables Included in Model)	R^2	Adj. R^2	P
Small Scale Plant Species Richness				
Restored Wetlands	0.892 – 0.347(AREA) – 0.002(WT)	45.3	41.5	<0.001
Waterbird Species Richness				
Breeding Season	-0.548 + 0.416(AREA) + 0.031(LSVEG) - 0.004(HP)	78.2	72.7	<0.001
Fall Migration	0.273 + 0.514(AREA) - 0.037(VS) + 0.003(WT)	81.3	76.6	<0.001
Spring Migration	-0.888 + 0.339(AREA) + 0.04(LSVEG) + 1.028(FQAI) + 0.005(WT) - 0.005(HP)	74.8	62.2	0.008

Table 5. Spearman's rho and p-values for breeding season waterbird richness (BSBIRD) correlation with HGM functional indices in riverine wetlands along the Deep Fork River, Oklahoma, USA (n = 16). LSVEG = plant species richness determined from transects, FQAI = floristic quality assessment index, VS = vertical structure of the plant community, HP = hydroperiod during the breeding season, WT = median water table during the breeding season, SURRWL = % of the surrounding landscape that is wetland. * indicates significance at $P \leq 0.10$. † indicates a strong relationship at $\rho \geq 0.50$ for significant p-values.

	ρ	P
LSVEG	0.685†	0.003*
FQAI	0.415	0.110
VS	0.229	0.394
OPEN	-0.010	0.970
ANNUAL	0.010	0.970
HP	0.588†	0.017*
WT	0.675†	0.004*
SURRWL	0.327	0.217

Table 6. Spearman's rho and p-values for fall migration waterbird richness (FMBIRD) correlation with HGM functional indices in riverine wetlands along the Deep Fork River, Oklahoma, USA (n = 16). LSVEG = plant species richness determined from transects, FQAI = floristic quality assessment index, VS = vertical structure of the plant community, HP = hydroperiod during fall migration, WT = median water table during fall migration, SURRWL = % of the surrounding landscape that is wetland. * indicates significance at $P \leq 0.10$. † indicates a strong relationship at $\rho \geq 0.50$ for significant p-values.

	ρ or *r*[a]	P
LSVEG	0.521†	0.039*
FQAI	-0.078	0.774
VS	-0.110	0.685
OPEN	0.236	0.380
ANNUAL	0.262	0.327
HP	0.553†	0.026*
WT	0.485	0.057*
SURRWL	0.219	0.414

[a] LSVEG, FQAI, and VS correlations were analyzed using Pearson correlation. HP, WT, and SURRWL correlations were analyzed using Spearman correlation.

Table 7. Spearman's rho and p-values for spring migration waterbird richness (SMBIRD) correlation with HGM functional indices in riverine wetlands along the Deep Fork River, Oklahoma, USA (n = 16). LSVEG = plant species richness determined from transects, FQAI = floristic quality assessment index, VS = vertical structure of the plant community, HP = hydroperiod during spring migration, WT = median water table during spring migration, SURRWL = % of the surrounding landscape that is wetland. * indicates significance at $P \leq 0.10$. † indicates a strong relationship at $\rho \geq 0.50$ for significant p-values.

	ρ or *r*[a]	P
LSVEG	0.554†	0.026*
FQAI	0.193	0.473
VS	0.061	0.821
OPEN	0.261	0.329
ANNUAL	0.301	0.258
HP	0.303	0.253
WT	0.427	0.099*
SURRWL	0.330	0.211

[a] LSVEG, FQAI, and VS correlations were analyzed using Pearson correlation. HP, WT, and SURRWL correlations were analyzed using Spearman correlation.

APPPENDICES

APPENDIX 1. LIST OF SCIENTIFIC AND COMMON NAMES FOR ALL WATERBIRD AND PLANT SPECIES OBSERVED IN THE STUDY.

Table 1.1. List of scientific and common names for all waterbird species observed WRP and natural riverine wetlands along the Deep Fork River, Oklahoma, USA and in which treatment(s) and seasons(s) they were observed. B = observed during the breeding season (June – early July 2009). F = observed during fall migration (mid-August – early December 2009). S = observed during spring migration (March – early May 2010).

Scientific Name	Common Name	Natural Presence	WRP Presence
Actitis macularia	Spotted sandpiper		B
Agelaius phoeniceus	Red-winged blackbird	B, F, S	B, F, S
Aix sponsa	Wood duck	B, F, S	B, F, S
Anas acuta	Northern pintail	F, S	F
Anas americana	American wigeon	F	F
Anas clypeata	Northern shoveler	F, S	F, S
Anas crecca	Green-winged teal	F, S	F, S
Anas discors	Blue-winged teal	F, S	B, F, S
Anas platyrhynchos	Mallard	F, S	B, F, S
Anas strepera	Gadwall	F, S	F, S
Ardea alba	Great egret	B, F, S	B, F, S
Ardea herodias	Great blue heron	B, F, S	B, F, S
Aythya affinis	Lesser scaup	S	F
Aythya americana	Redhead		F
Aythya collaris	Ring-necked duck	S	
Aythya marila	Greater scaup		B
Botaurus lentiginosus	American bittern	S	B, S
Branta canadensis	Canada goose	F, S	B, F, S
Bubulcus ibis	Cattle egret	B	B, F
Bucephala albeola	Bufflehead	F	
Butorides virescens	Green heron	B, F	B
Calidris bairdii	Baird's sandpiper		F
Calidris melanotos	Pectoral sandpiper		S
Calidris spp.	Shorebird spp.		B, F, S
Ceryle alcyon	Belted kingfisher	B, F	B, F
Charadrius vociferus	Killdeer	B	B, F, S
Cistothorus palustris	Marsh wren	F	
Dendrocygna autumnalis	Black-bellied whistling-duck	B	B
Egretta caerulea	Little blue heron	B, F	B, F, S
Egretta thula	Snowy egret	B	B, F, S
Eudocimus albus	White ibis	F	B
Fulica americana	American coot	F, S	B, F, S
Gallinago delicata	Wilson’s snipe	F	F, S
Geothlypis trichas	Common yellowthroat	B, S	B, S
Larus spp.	Gull spp.		F
Nyctanassa violacea	Yellow-crowned night-heron	B	B, F
Pelecanus erythrorhynchos	American white pelican		S

Table 1.1 cont.			
Phalacrocorax auritus	Double-crested cormorant	S	F, S
Plegadis chihi	White-faced ibis	F	B, F, S
Pluvialis squatarola	Black-bellied plover		F
Podilymbus podiceps	Pied-billed grebe	F, S	F, S
Porzana carolina	Sora		F, S
Protonotaria citrea	Prothonotary warbler	B	B
Rallus limicola	Virginia rail		B
Tringa flavipes	Lesser yellowlegs		S
Tringa melanoleuca	Greater yellowlegs	S	S
Tringa solitaria	Solitary sandpiper	F	F

Table 1.2. List of scientific and common names for all plant species observed WRP and natural riverine wetlands along the Deep Fork River, Oklahoma, USA and in which treatment(s) and year(s) they were observed. 1 = observed in 2009. 2 = observed in 2010.

Scientific Name	Common Name	Natural Presence	WRP Presence
Amaranthus tuberculatus (Moq.) Sauer	roughfruit amaranth	2	1, 2
Ambrosia psilostachya DC.	western ragweed		1
Ambrosia trifida L.	giant ragweed		1,2
Ammannia coccinea Rottb.	valley redstem	1	1, 2
Andropogon virginicus L.	broomsedge bluestem		1
Apocynum cannabinum L.	Indianhemp	2	
Azolla caroliniana Willd.	Carolina mosquitofern	1, 2	
Bacopa rotundifolia (Michx.) Wettst.	disk waterhyssop		2
Bidens spp.	beggartick spp.	1	
Campsis radicans (L.) Seem. ex Bureau	trumpet vine	1, 2	2
Cardiospermum halicacabum L.	balloon vine	1, 2	1, 2
Carex crus-corvi Shuttlw. ex Kunze	ravenfoot sedge	1	
Carex lupulina Muhl. ex Willd.	hop sedge	1, 2	
Carex tribuloides Wahlenb.	blunt broom sedge	1	2
Carex vulpinoidea Michx.	fox sedge	1, 2	1, 2
Carya illinoinensis (Wangenh.) K. Koch	pecan		1, 2
Cephalanthus occidentalis L.	common buttonbush	1, 2	1, 2
Ceratophyllum demersum L.	coontail	1, 2	1, 2
Chasmanthium latifolium (Michx.) Yates	fish-on-a-fishing-pole	2	
Chloris verticillata Nutt.	windmill grass		1
Conyza canadensis (L.) Cronquist	marestail	1	1, 2
Coreopsis tinctoria Nutt.	plains coreopsis		1
Cuscuta spp.	dodder		1, 2
Cynodon dactylon (L.) Pers.	bermudagrass	1	1, 2
Cyperaceae spp.	sedge spp.		1, 2
Cyperus erythrorhizos Muhl.	redroot flatsedge	2	1
Cyperus odoratus L.	fragrant flatsedge	1	1, 2
Desmanthus illinoensis (Michx.) MacMill. ex B.L. Rob. & Fernald	prairie bundleflower		1, 2
Diodia virginiana L.	Virginia buttonweed		2
Echinochloa crus-galli (L.) P. Beauv.	barnyardgrass	1, 2	1, 2
Echinochloa muricata (P. Beauv.) Fernald	rough barnyardgrass		1
Echinodorus berteroi (Spreng.) Fassett	upright burhead		1, 2
Echinodorus cordifolius (L.) Griseb.	creeping burhead	2	
Eclipta prostrata (L.) L.	false daisy	1	2
Eleocharis macrostachya Britton	pale spikerush	1	1, 2
Eleocharis obtusa (Willd.) Schult.	blunt spikerush		1
Eleocharis palustris (L.) Roem. & Schult.	common spikerush	1	

Table 1.2 cont.			
Eleocharis parvula (Roem. & Schult.) Link ex Bluff, Nees & Schauer	dwarf spikerush		2
Euphorbia spp.	spurge spp.		1, 2
Fraxinus americana L.	white ash	1	
Fraxinus pennsylvanica Marsh.	green ash		2
Gratiola neglecta Torr.	clammy hedgehyssop		1
Helianthus annuus L.	common sunflower		1
Heliotropium curassavicum L.	salt heliotrope	1, 2	1, 2
Heliotropium indicum L.	Indian heliotrope	2	2
Hibiscus laevis All.	halberdleaf rosemallow	1, 2	1, 2
Hibiscus trionum L.	flower of an hour		1
Ipomoea lacunosa L.	whitestar		1, 2
Iva annua L.	sumpweed	1, 2	1, 2
Justicia americana (L.) Vahl	American water-willow		1, 2
Lamiaceae spp..	mint family	1	1
Leersia oryzoides (L.) Sw.	rice cutgrass	1, 2	1, 2
Lemna minor L.	common duckweed	1	2
Lespedeza cuneata (Dum. Cours.) G. Don	sericea lespedeza		1, 2
Lespedeza procumbens Michx.	trailing lespedeza		1
Lindernia dubia (L.) Pennell	slender false pimpernel		1, 2
Ludwigia peploides (Kunth) P.H. Raven	floating primrose-willow	1, 2	1, 2
Lythrum alatum Pursh	winged loosestrife		1
Najas guadalupensis (Spreng.) Magnus	southern waternymph	1	1, 2
Neeragrostis reptans (Michx.) Nicora	creeping lovegrass	1	1
Nelumbo lutea Willd.	American lotus	1, 2	1, 2
Oxalis spp.	woodsorrel spp.		1
Panicum capillare L.	witchgrass		1
Panicum virgatum L.	switchgrass		1, 2
Paspalidium geminatum (Forssk.) Stapf	Egyptian panicgrass		2
Paspalum dilatatum Poir.	dallisgrass		2
Paspalum distichum L.	knotgrass	1, 2	1, 2
Paspalum floridanum Michx.	Florida paspalum		1
Paspalum pubiflorum Rupr. ex Fourn.	hairyseed paspalum		2
Phyla lanceolata (Michx.) Greene	lanceleaf frogfruit	1, 2	1, 2
Physalis angulata L.	cutleaf groundcherry		1, 2
Pluchea odorata (L.) Cass.	salt-marsh fleabane	1	1
Polygonum amphibium L.	water smartweed	1, 2	1, 2
Polygonum aviculare L.	prostrate knotweed	2	1, 2
Polygonum hydropiperoides Michx.	swamp smartweed	1, 2	1, 2
Polygonum lapathifolium L.	nodding smartweed	1	1, 2
Polygonum pensylvanicum L.	Pennsylvania smartweed	1	1, 2
Populus deltoides Bartram ex Marsh.	eastern cottonwood		1, 2

Table 1.2 cont.			
Potamogeton nodosus Poir.	longleaf pondweed	1, 2	1, 2
Rhynchospora corniculata (Lam.) A. Gray	shortbristle horned beaksedge	1, 2	1, 2
Rubus spp.	blackberry spp.		2
Rumex crispus L.	curly dock	1, 2	1, 2
Sagittaria latifolia Willd.	broadleaf arrowhead		1, 2
Sagittaria platyphylla (Engelm.) J.G. Sm.	delta arrowhead	1, 2	1, 2
Salix nigra Marsh.	black willow	1, 2	1, 2
Samolus valerandi L. ssp. parviflorus (Raf.) Hultén	seaside brookweed		1
Schoenoplectus acutus (Muhl. ex Bigelow) A. Löve & D. Löve	hardstem bulrush		1
Schoenoplectus americanus (Pers.) Volkart ex Schinz & R. Keller	American bulrush	2	1, 2
Schoenoplectus tabernaemontani (C.C. Gmel.) Palla	softstem bulrush	1, 2	
Sesbania herbacea (Mill.) McVaugh	bigpod sesbania		1, 2
Setaria viridis (L.) P. Beauv.	green foxtail		1, 2
Solidago canadensis L.	Canada goldenrod		1, 2
Spirodela polyrrhiza (L.) Schleid.	common duckmeat	1, 2	1, 2
Sporobolus heterolepis (A. Gray) A. Gray	prairie dropseed	1	1
Symphyotrichum dumosum (L.) G.L. Nesom	bushy aster		1
Symphyotrichum subulatum (Michx.) G.L. Nesom	eastern annual saltmarsh aster	1	1, 2
Tamarix ramosissima Ledeb.	saltcedar	1, 2	
Teucrium canadense L.	Canada germander		1
Thalia dealbata Fraser ex Roscoe	powdery alligator-flag	1, 2	
Toxicodendron radicans (L.) Kuntze	poison ivy	2	2
Typha domingensis Pers.	southern cattail	1, 2	1, 2
Typha latifolia L.	broadleaf cattail	1, 2	
Utricularia gibba L.	humped bladderwort	1	
Vitis spp.	grape spp.		1
Wolffia columbiana Karst.	Columbian watermeal	1, 2	
Xanthium strumarium L.	rough cocklebur	1, 2	1, 2
Zizaniopsis miliacea (Michx.) Döll & Asch.	giant cutgrass	1, 2	1, 2
	unknown forb	1	1, 2
	unknown gramminoid		2

APPENDIX 2. MEAN, MINIMUM, AND MAXIMUM VALUES OBSERVED FOR EACH FUNCTIONAL INDEX USED TO TEST FOR RELATIONSHIP WITH PLANT AND WATERBIRD SPECIES RICHNESS.

Table 2.1. Summary of the minimum, maximum, and mean or median for functional indices tested for relationship with large and small scale plant species richness in natural and restored wetlands along the Deep Fork River, Oklahoma, USA. SOM = soil organic matter, TN = total carbon, P = available phosphorus, EC = electrical conductivity, WT = median growing season water table, and SAT = percent of the growing season the rooting zone is saturated.

	SOM (%)		
	Mean	Minimum	Maximum
Large Scale			
Natural Wetlands	3.91	2.27	5.93
Restored Wetlands	2.80	1.93	3.35
Small Scale			
Natural Wetlands	3.93	2.07	6.87
Restored Wetlands	2.80	1.65	5.10
	TN (%)		
	Mean	Minimum	Maximum
Large Scale			
Natural Wetlands	0.21	0.13	0.29
Restored Wetlands	0.16	0.12	0.19
Small Scale			
Natural Wetlands	0.21	0.12	0.32
Restored Wetlands	0.16	0.11	0.26
	P (ppm)		
	Mean	Minimum	Maximum
Large Scale			
Natural Wetlands	19.03	7.13	52.31
Restored Wetlands	17.41	5.88	27.44
Small Scale			
Natural Wetlands	18.73	4.50	67.25
Restored Wetlands	17.41	4.50	53.50
	EC (dS/m)		
	Mean	Minimum	Maximum
Large Scale			
Natural Wetlands	2.9	0.7	11.8
Restored Wetlands	2.0	0.8	7.3
Small Scale			
Natural Wetlands	2.1	0.6	8.0
Restored Wetlands	2.0	0.6	11.9
	pH		
	Mean	Minimum	Maximum
Large Scale			
Natural Wetlands	6.73	5.07	8.03
Restored Wetlands	7.21	6.20	8.03
Small Scale			
Natural Wetlands	6.70	4.90	8.23
Restored Wetlands	7.21	6.17	8.27

Table 2.1. cont.

	WT (cm)		
	Median	Minimum	Maximum
Large Scale			
Natural Wetlands	-6.75	-83.88	23.88
Restored Wetlands	-33.27	-96.63	19.00
Small Scale			
Natural Wetlands	1.50	-87.50	34.25
Restored Wetlands	-48.00	-105.00	73.50

Table 2.1 cont.

	SAT (%)		
	Mean	Minimum	Maximum
Large Scale			
Natural Wetlands	74.51	30.00	100.00
Restored Wetlands	55.52	28.13	85.00
Small Scale			
Natural Wetlands	75.13	20.00	100.00
Restored Wetlands	55.52	0.00	100.00

Appendix 2.2. Summary of the minimum, maximum, and mean or median for functional indices tested for relationship with waterbird species richness during the breeding season, fall migration, and spring migration in riverine wetlands along the Deep Fork River, Oklahoma, USA. LSVEG = plant species richness, VS = vertical structure of the plant community, FQAI = floristic quality assessment index, HP = hydroperiod for each season, WT = median water table for each season, and SURRWL = percent of the surrounding landscape that is wetland.

	LSVEG		
	Mean	Minimum	Maximum
All Seasons	14.53	8.00	23.50
	VS (dm)		
	Mean	Minimum	Maximum
All Seasons	7.12	0.85	14.10
	FQAI		
	Mean	Minimum	Maximum
All Seasons	0.85	0.49	1.22
	HP (%)		
	Mean	Minimum	Maximum
Breeding Season	37.50	0.00	83.33
Fall Migration	57.50	6.25	100.00
Spring Migration	75.78	25.00	100.00
	WT (cm)		
	Median	Minimum	Maximum
Breeding Season	-46.13	-105	16.63
Fall Migration	2.94	-79.63	41.50
Spring Migration	29.38	-22.13	96.25
	SURRWL (%)		
	Mean	Minimum	Maximum
All Seasons	17.64	2.62	32.68

APPENDIX 3. THE CORRELATIONS BETWEEN FUNCTIONAL INDICES AND PLANT SPECIES RICHNESS MEASURED ON A SMALL AND LARGE SCALE AS WELL AS WATERBIRD SPECIES RICHNESS MEASURED DURING THE BREEDING SEASON, FALL MIGRATION, AND SPRING MIGRATION.

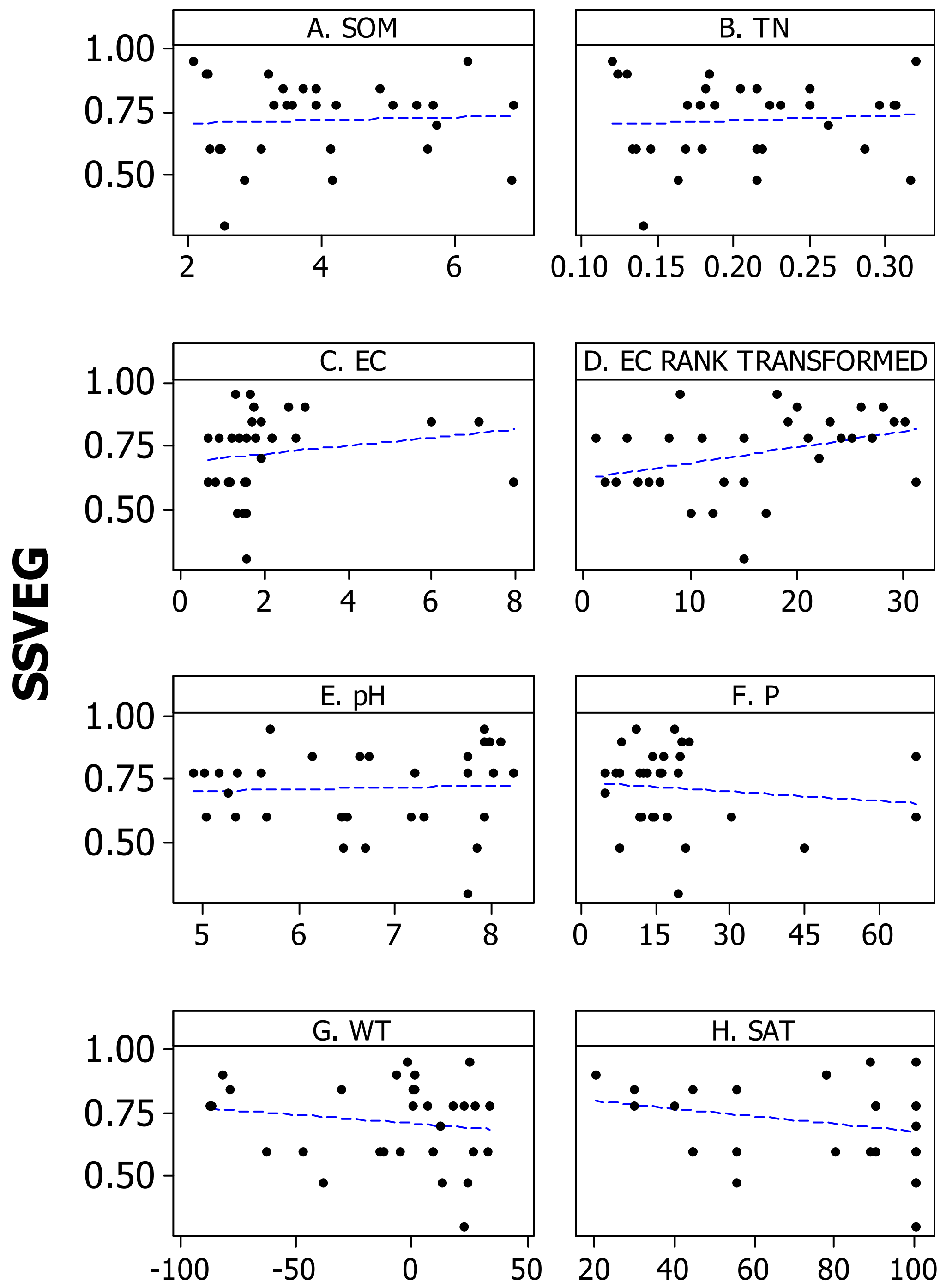

Figure 1. The scatterplot and correlation line between functional indices and small scale plant species richness (SSVEG) in natural riverine wetlands along the Deep Fork River, Oklahoma, USA (n = 32). Graph A: soil organic matter (SOM; %), Graph B: total nitrogen (TN; %), Graph C: electrical conductivity (EC; dS/m), Graph D: rank transformed electrical conductivity, Graph E: pH, Graph F: available phosphorus (P; ppm), Graph G: median growing season water table (WT; cm), Graph H: percent of the growing season the rooting zone was saturated (SAT; %).

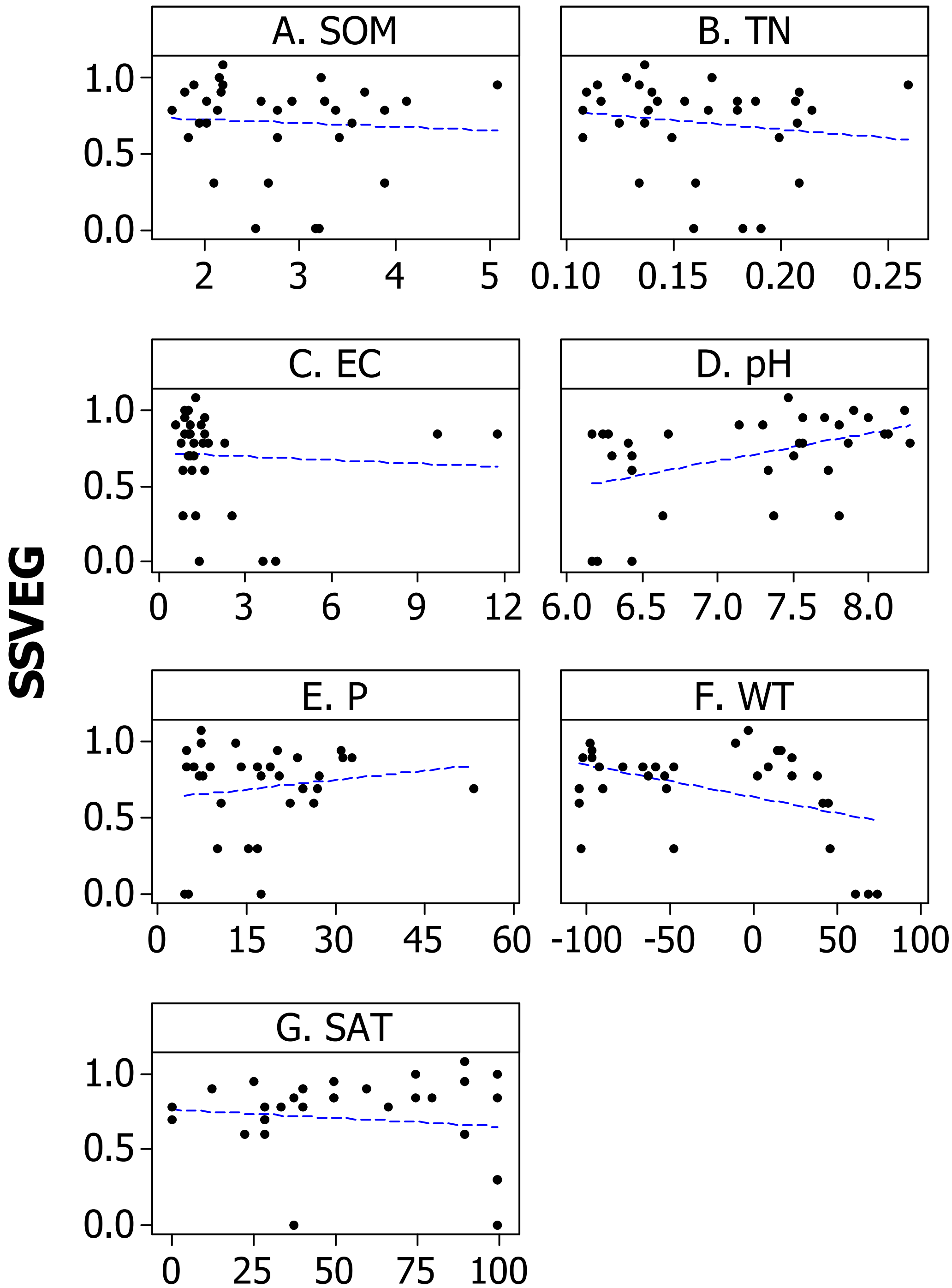

Figure 2. The scatterplot and correlation line between functional indices and small scale plant species richness (SSVEG) in restored riverine wetlands along the Deep Fork River, Oklahoma, USA (n = 32). Graph A: soil organic matter (SOM; %), Graph B: total nitrogen (TN; %), Graph C: electrical conductivity (EC; dS/m), Graph D: pH, Graph E: available phosphorus (P; ppm), Graph f: median growing season water table (WT; cm), Graph G: percent of the growing season the rooting zone was saturated (SAT; %).

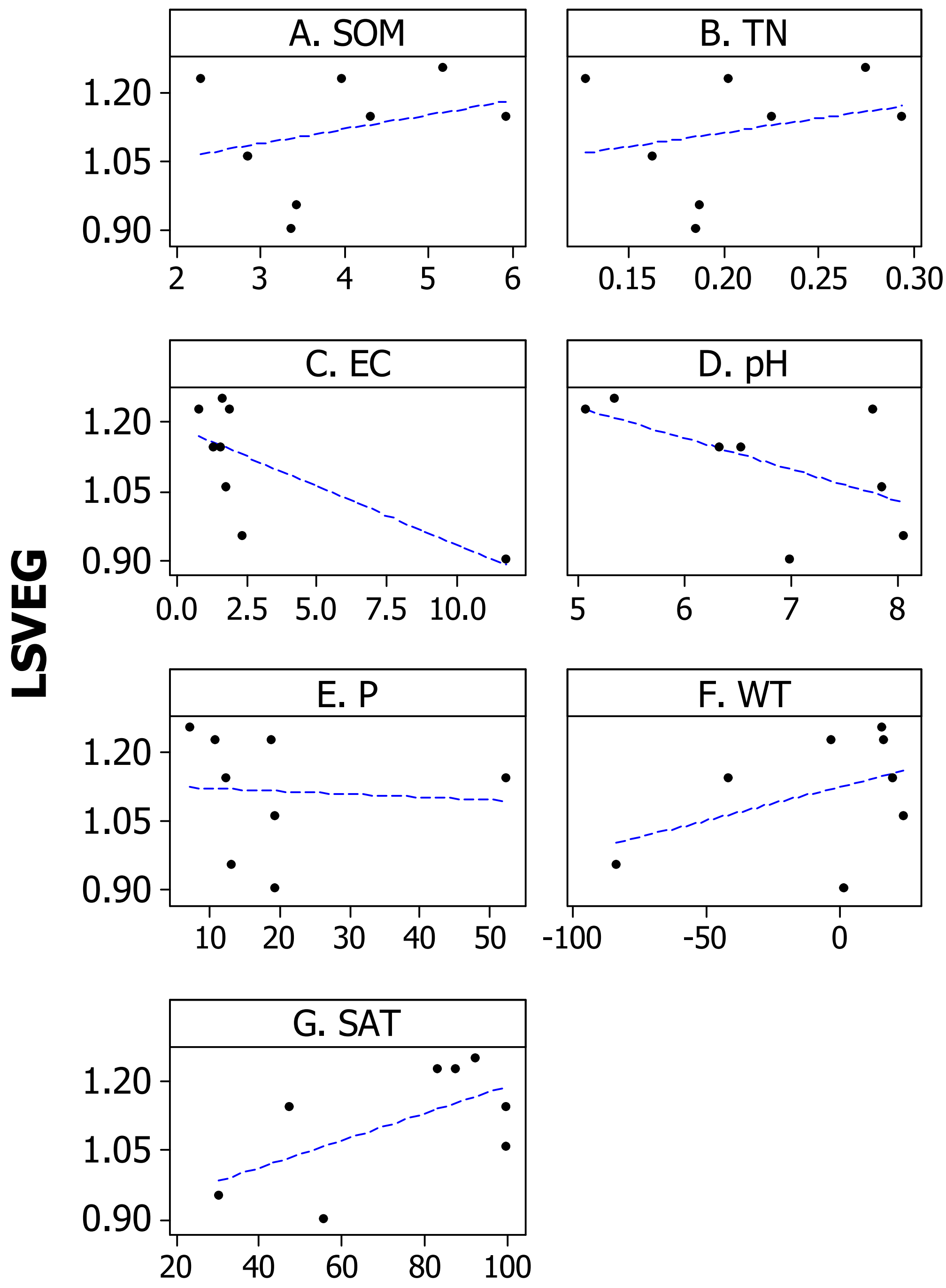

Figure 3. The scatterplot and correlation line between functional indices and large scale plant species richness (LSVEG) in natural riverine wetlands along the Deep Fork River, Oklahoma, USA (n = 32). Graph A: soil organic matter (SOM; %), Graph B: total nitrogen (TN; %), Graph C: electrical conductivity (EC; dS/m), Graph D: pH, Graph E: available phosphorus (P; ppm), Graph f: median growing season water table (WT; cm), Graph G: percent of the growing season the rooting zone was saturated (SAT; %).

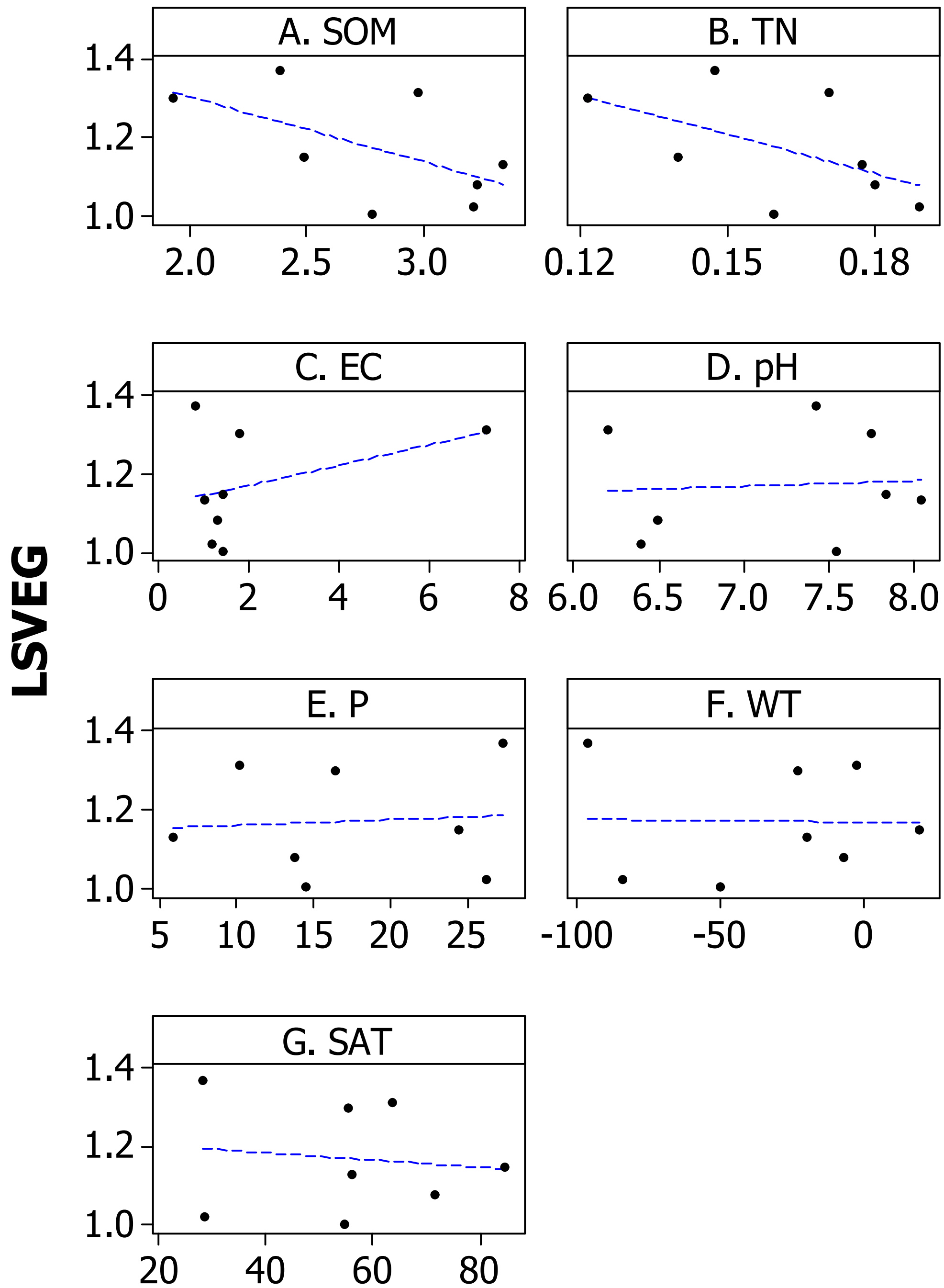

Figure 4. The scatterplot and correlation line between functional indices and large scale plant species richness (LSVEG) in restored riverine wetlands along the Deep Fork River, Oklahoma, USA (n = 32). Graph A: soil organic matter (SOM; %), Graph B: total nitrogen (TN; %), Graph C: electrical conductivity (EC; dS/m), Graph D: pH, Graph E: available phosphorus (P; ppm), Graph f: median growing season water table (WT; cm), Graph G: percent of the growing season the rooting zone was saturated (SAT; %).

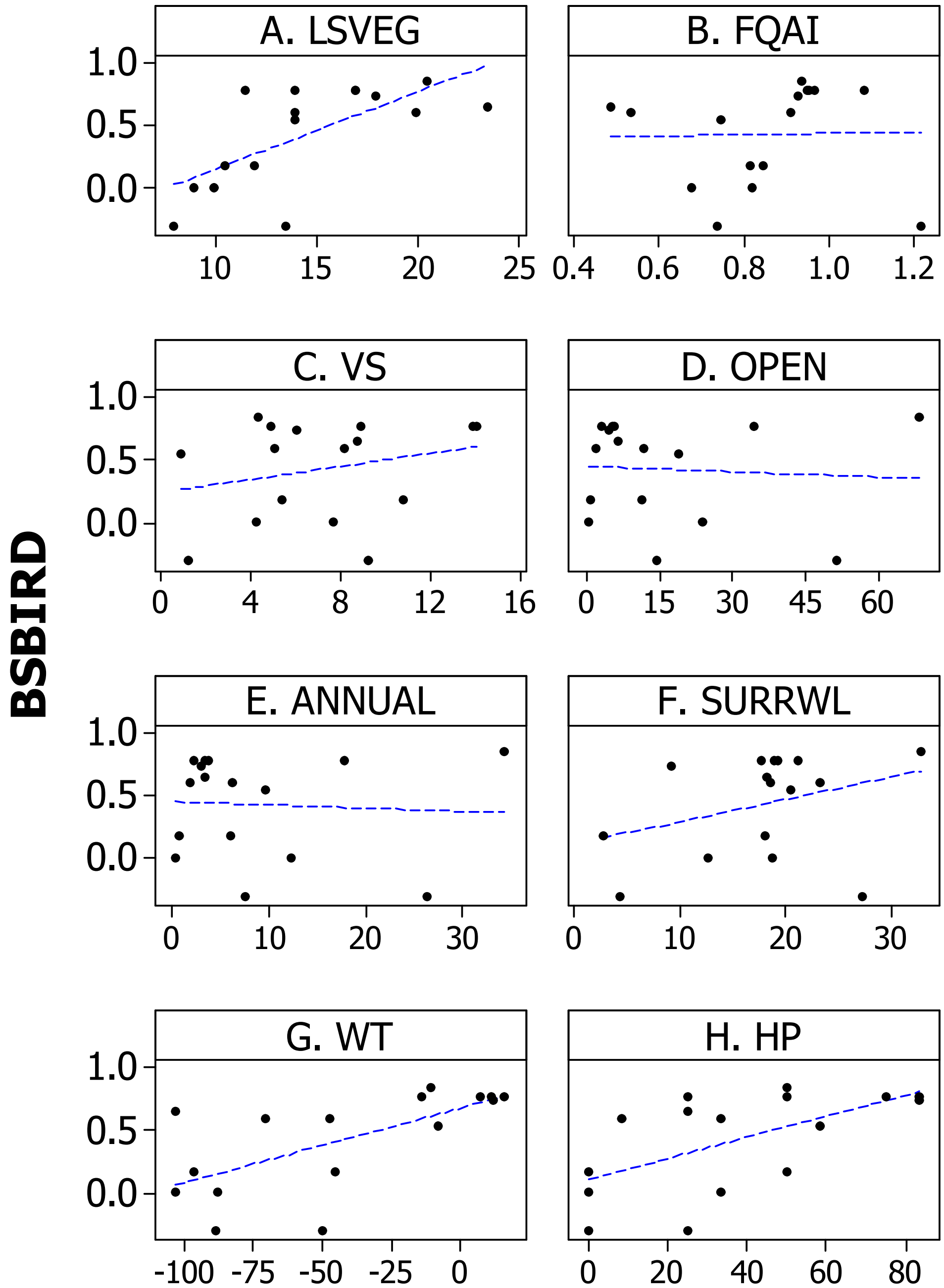

Figure 5. The scatterplot and correlation line between functional indices and breeding season waterbird species richness (BSBIRD) in riverine wetlands along the Deep Fork River, Oklahoma, USA (n = 16). Graph A: large scale plant species richness (LSVEG), Graph B: floristic quality assessment index (FQAI), Graph C: vertical structure of the plant community (VS; dm), Graph D: proportion of the wetland that is open water or bare ground (OPEN; %), Graph E: proportion of the plants that are annual (ANNUAL; %), Graph F: proportion of the surrounding 3 km that is wetland (SURRWL; %), Graph G: median water table during the breeding season (WT; cm), Graph H: hydroperiod during the breeding season (HP; %).

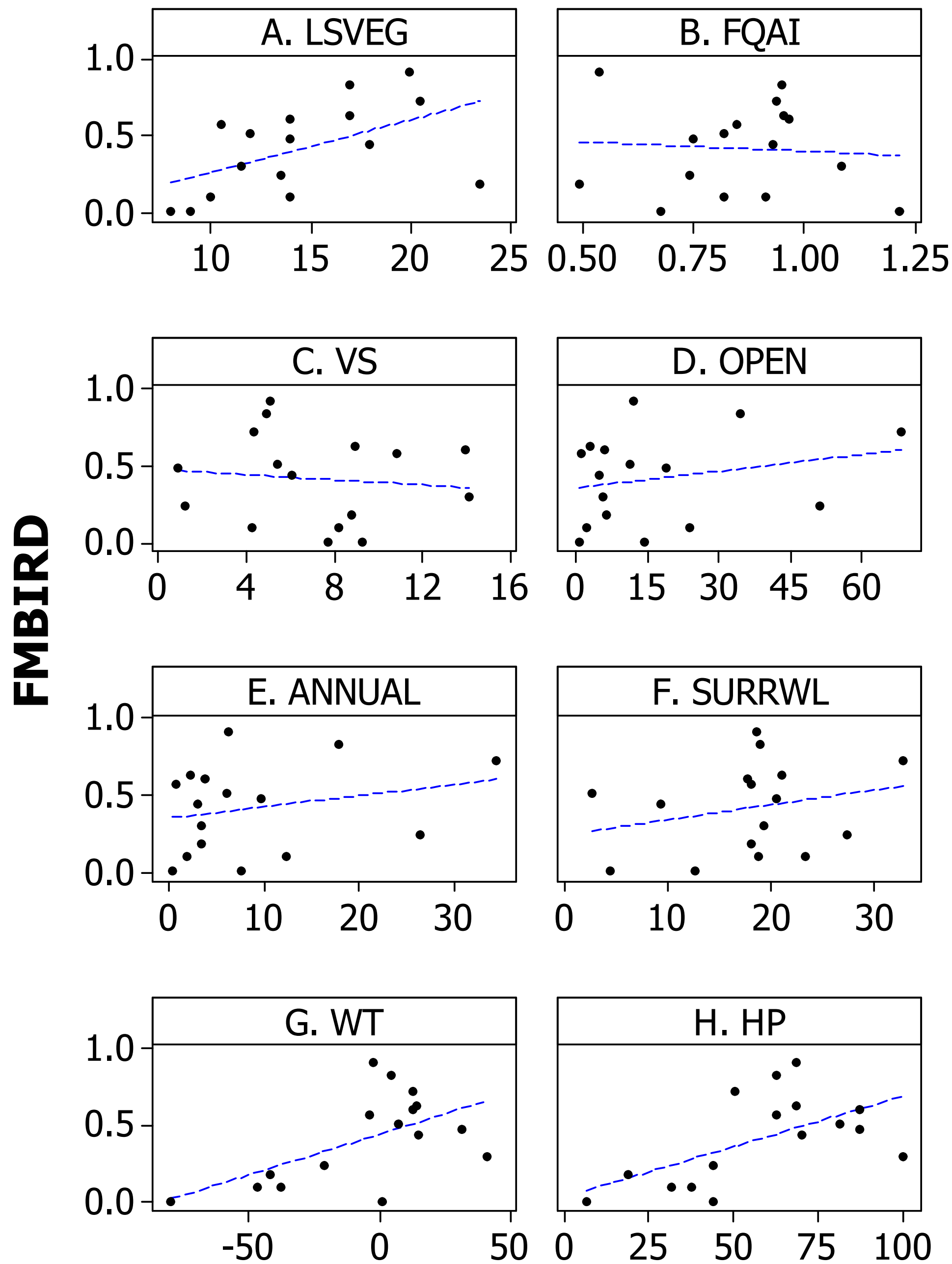

Figure 6. The scatterplot and correlation line between functional indices and fall migration waterbird species richness (FMBIRD) in riverine wetlands along the Deep Fork River, Oklahoma, USA (n = 16). Graph A: large scale plant species richness (LSVEG), Graph B: floristic quality assessment index (FQAI), Graph C: vertical structure of the plant community (VS; dm), Graph D: proportion of the wetland that is open water or bare ground (OPEN; %), Graph E: proportion of the plants that are annual (ANNUAL; %), Graph F: proportion of the surrounding 3 km that is wetland (SURRWL; %), Graph G: median water table during the breeding season (WT; cm), Graph H: hydroperiod during the breeding season (HP; %).

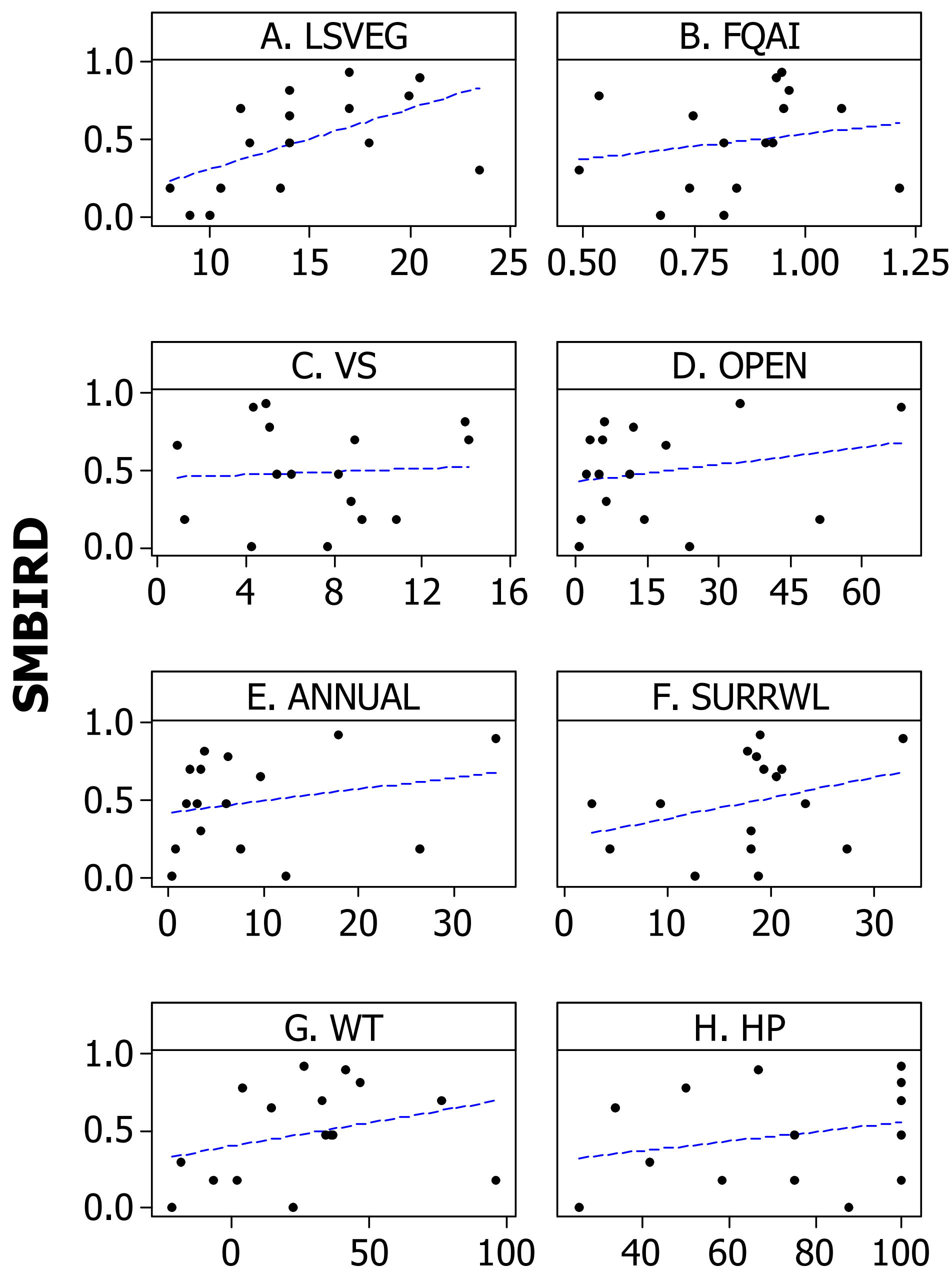

Figure 7. The scatterplot and correlation line between functional indices and spring migration waterbird species richness (SMBIRD) in riverine wetlands along the Deep Fork River, Oklahoma, USA (n = 16). Graph A: large scale plant species richness (LSVEG), Graph B: floristic quality assessment index (FQAI), Graph C: vertical structure of the plant community (VS; dm), Graph D: proportion of the wetland that is open water or bare ground (OPEN; %), Graph E: proportion of the plants that are annual (ANNUAL; %), Graph F: proportion of the surrounding 3 km that is wetland (SURRWL; %), Graph G: median water table during the breeding season (WT; cm), Graph H: hydroperiod during the breeding season (HP; %).

VITA

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Candidate for the Degree of

Master of Science

Thesis: ASSESSING IMPORTANT BIOLOGICAL FUNCTIONS PROVIDED BY THE WETLANDS RESERVE PROGAM IN OKLAHOMA, USA

Major Field: Natural Resource Ecology and Management

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Institution: Oklahoma State University Location: Stillwater, Oklahoma

Title of Study: ASSESSING IMPORTANT BIOLOGICAL FUNCTIONS PROVIDED BY THE WETLANDS RESERVE PROGAM IN OKLAHOMA, USA

Pages in Study: 141 Candidate for the Degree of Master of Science

Major Field: Natural Resource Ecology and Management

Scope and Method of Study: The objectives of this study were to (1) evaluate Wetlands Reserve Program (WRP) wetlands to assess whether they are performing the functions “maintaining plant communities,” “maintaining waterbird communities,” and “maintaining interspersion” similarly to natural wetlands and (2) to make certain the functional indices used in hydrogeomorphic (HGM) models are related to a direct measure of species richness for the functions “maintaining plant communities” and “maintaining waterbird communities.” Study sites included 8 naturally occurring wetlands and 8 hydrologically managed WRP wetlands. All sites were herbaceous, riverine wetlands of the Deep Fork River, OK. For Objective 1, waterbird abundance, richness, evenness, and diversity were used to assess waterbird communities; plant species richness, evenness and diversity were used to assess plant communities; and the area of wetland habitat within 3 km of study sites was used to assess interspersion. Waterbird and plant metrics were determined from field surveys. Wetland area within 3 km was determined using National Wetland Inventory maps. For Objective 2, plant and waterbird species richness were tested for relationships with potential functional indices used in HGM models.

Findings and Conclusions: When comparing WRP to natural wetlands, waterbird metrics, plant species evenness, and interspersion were similar. However, plant species richness and diversity were higher in WRP than natural wetlands. Overall, these WRP wetlands were providing similar waterbird communities, plant communities, and interspersion to natural wetlands. When testing for relationships between species richness and HGM functional indices, plant species richness was related to electrical conductivity, pH, duration of rooting zone saturation, and growing season water table. Due to weak relationships between functional indices and plant species richness, HGM assessment was determined to be an unsuccessful measurement of “maintaining plant communities.” Waterbird species richness was related to plant species richness, hydroperiod, water table, vertical structure, and floristic quality assessment index. Due to strong relationships between functional indices and waterbird species richness, HGM was determined to be successful at measuring “maintaining waterbird communities.”

ADVISER’S APPROVAL: Dr. Craig A. Davis