NORTH DAKOTA WETLANDS: CHANGES OVER FIVE YEARS IN PRAIRIE POTHOLE WETLANDS AND A DESCRIPTION OF VEGETATIVE AND SOIL PROPERTIES IN A NORTH DAKOTA FEN

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MASTER OF SCIENCE

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ABSTRACT

Wetlands provide ecosystem services such as water quality, flood attenuation, primary productivity, biodiversity, and provide habitats for wildlife. Land use conversions from natural to agricultural and urban landscapes threaten the quantity and quality of wetlands globally. Monitoring remaining wetlands has become increasingly important as degradation persists, particularly in agriculturally productive regions like the Prairie Pothole Region (PPR) of North Dakota. Two studies were completed in the summers of 2016 and 2017 to contribute to efforts to monitor wetlands in North Dakota.

The first study aimed to assess the overall condition of prairie pothole wetlands across North Dakota using the North Dakota Rapid Assessment Method. Data from 2016 was compared to similar data collected in 2011 to determine how wetland condition has changed between the two years. The second study aimed to describe the vegetative and soil properties in a natural fen in central North Dakota.

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CHAPTER 1: CHANGES OVER FIVE YEARS IN WETLAND CONDITION IN THE PRAIRIE POTHOLE REGION OF NORTH DAKOTA USING A RAPID ASSESSMENT METHOD

Abstract

Wetlands are valuable for water quality, flood attenuation, primary productivity, biodiversity, and wildlife habitats. Approximately half the hectares of wetlands in North Dakota have been lost due to anthropic disturbances, particularly land use conversions for agricultural production and urban development. As wetland degradation continues, condition assessments of the remaining wetlands become increasingly important to monitor ecosystem quality. To assess the condition of prairie pothole wetlands in North Dakota, 44 pothole wetlands were assessed in 2011 and 39 pothole wetlands were assessed in 2016 using the North Dakota Rapid Assessment Method. Results from the 2011 assessment were compared to the data collected in 2016. The data show temporary and seasonal wetlands had no significant difference in condition between 2011 and 2016, whereas semipermanent wetlands had significantly lower overall condition in 2016. This study is part of an ongoing effort to document the condition of wetland resources in North Dakota.

Literature Review

Prairie Pothole Region

The Prairie Pothole Region (PPR), within the grasslands of the Northern Great Plains, is characterized by high concentrations of depressional wetlands extending from central Alberta to central Iowa (Figure 1.1) (Guntenspergen et al. 2002; Niemuth et al. 2010; Gleason et al. 2011; Dahl 2014). This region, covering approximately 777,000 square kilometers of North America (Rosen et al. 1995; Dahl 2014; Mushet 2016), including 274,500 square kilometers in the United

States (Rosen et al. 1995), is spotted with millions of pothole wetlands formed by glacial activity during the late Wisconsinan glaciation (Sloan 1970; Gilbert et al. 2006; Dahl 2014; Mushet 2016). The topographic depressions were formed by repeated glacial advancement and retreat within the region's cold dry climate (Conly & van der Kamp 2001; van der Kamp et al. 2016). Prairie potholes, though individually small, represent one of the most hydrologically diverse inland wetland systems in North America (Gleason et al. 2011). Across the PPR, wetlands vary in size from small depressions, holding water for only a few days, to depressions that have permanent deep water (van der Kamp et al. 2016).

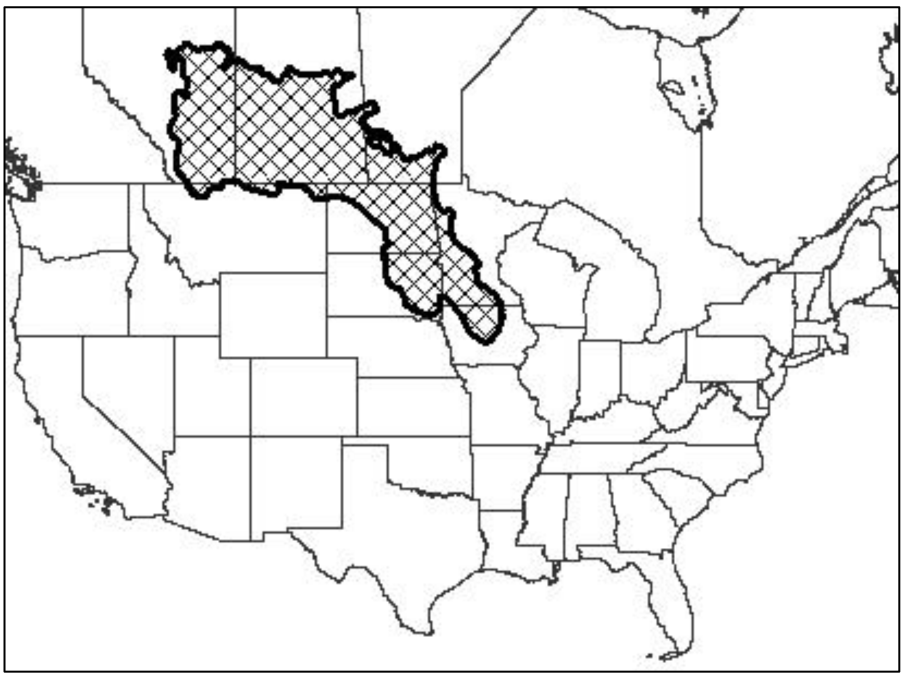

Figure 1.1. Map of the Prairie Pothole Region in North America.

The PPR has a semi-arid to sub-humid climate (van der Kamp et al. 2016) and is characterized by cyclic wet and dry periods (Guntenspergen et al. 2002; Dahl 2014; Cressey et al. 2016; Mushet 2016), where evapotranspiration exceeds precipitation (Mushet 2016; van der Kamp et al. 2016). The cyclic nature of precipitation in the PPR affects wetland condition (Cressey et al. 2016). The variable climatic conditions result in high biodiversity within prairie wetlands (Euliss & Mushet 2011), however variable weather conditions also make this region vulnerable to disturbances (Conly & van der Kamp 2001; Dahl 2014; Mushet 2016). Wetlands within the PPR are sensitive to the annual shifts in precipitation (Dahl 2014; van der Kamp et al. 2016), as biotic communities respond to the dynamic wet and dry periods (Cressey et al. 2016). A recent wet/dry cycle began in 1988 when the region experienced its most severe drought which lasted until the wet cycle began in 1993. The increased precipitation since 1993 has resulted in the merging of a number of wetlands in the PPR. These wetlands have more stable water levels, which affect the associated biotic communities (Cressey et al. 2016).

Wetland Functions

Wetlands provide ecosystem services such as filtering and storing water, improving water quality, preventing flooding, and protecting shorelines from erosion (Guntenspergen et al. 2002; Carletti et al. 2004; Dahl & Watmough 2007). Wetlands in the PPR are important in the maintenance of regional and national biodiversity, flood attenuation, nutrient cycling, carbon sequestration, and groundwater recharge (Guntenspergen et al. 2002; Gleason et al. 2011; Dahl 2014). Wetlands provide important habitat to water and grassland birds (Guntenspergen et al. 2002; Gleason et al. 2011; Dahl 2014), which rely on the PPR for breeding, nesting, and migration (Rosen et al. 1995; Gleason et al. 2011). Between 40 and 60 percent of North American waterfowl production occurs within the PPR, although this region represents only ten percent of the total breeding area in North America (Rosen et al. 1995; Guntenspergen et al. 2002). In addition to waterfowl, a number of mammal, amphibian, fish, algae, and plant species rely on wetland habitats (Rosen et al. 1995).

Hydrologic Classes in the PPR

According to the National Wetland Inventory (NWI), the majority of wetlands in the PPR are palustrine and lacustrine systems dominated by temporary, seasonal, and semipermanent water regimes (Gleason et al. 2011; Dahl 2014). Surface water is the dominant water input in temporary and seasonal wetlands (Euliss & Mushet 1996). In temporary wetlands, water loss occurs mostly through groundwater recharge, whereas in in seasonal wetlands water loss can occur through both groundwater recharge and evapotranspiration. Water inputs in semipermanent wetlands are a result of surface runoff and groundwater movement. Groundwater inputs stabilize water levels resulting in comparatively more permanent water supplies. Water loss in semipermanent wetlands is dominated by evapotranspiration (Euliss & Mushet 1996).

Temporary wetlands are defined by a wet-meadow zone dominating the deepest part of the wetland surrounded by a low prairie zone (i.e. upland vegetation) (Stewart & Kantrud 1971). Seasonal wetlands have shallow-marsh zones dominating the deepest part of the wetland and are surrounded by wet-meadow and low prairie zones. Semipermanent wetlands are classified as having the deepest part of the wetland dominated by a deep-marsh zone surrounded by shallow-marsh, wet-meadow, and low prairie zones (Stewart & Kantrud 1971). The hydrologic classification of a given wetland can change over time with extended periods of drought or increased precipitation, where changes in the vegetation zones may shift the classification of a given site (Stewart & Kantrud 1971, Euliss et al. 2004).

Seasonal and semipermanent water regimes predominate the total acreage of wetlands throughout the PPR. Although ephemeral and temporary wetlands are abundant, seasonal wetlands account for the greatest number of wetlands in the region (Stewart & Kantrud 1971, Hargiss 2009). In North Dakota, approximately 93 percent of the wetlands are classified as temporary and seasonal; however, these wetland types only account for 43 percent of the total water acreage in the state (DeKeyser et al. 2003).

Wetland Disturbances

Approximately 81 million hectares of wetlands existed in the continental United States prior to European settlement (USEPA 2002), with wetlands accounting for 16 to 18 percent of the land within the PPR (Dahl 1990; Mushet 2016). By the mid-1980s, wetland basins in the PPR had decreased by up to 65 percent, mostly due to drainage for agricultural production (Dahl 1990; Rosen et al. 1995; Euliss & Mushet 1996; Mushet 2016). In North Dakota, approximately two million hectares of wetlands existed prior to settlement, but by 1984, less than half of those hectares remained (Leitch & Baltezore 1992; Dahl 2014).

The PPR is one of the most productive agricultural regions in the world (Gilbert et al. 2006; Gleason et al. 2011; Dahl 2014), responsible for one-third of the annual production of wheat, corn, barley, and soybeans within the U.S. (Gleason et al. 2011; Dahl 2014). The wet and dry cycles of the region increase primary productivity and elemental cycling making the landscape prone to agricultural conversion (Gilbert et al. 2006). The productivity potential of the region has resulted in the loss of native grassland and wetland habitats, with greater than 50 percent lost through the conversion to agriculture (Gleason et al. 2011). Anthropic disturbances including the drainage of wetland basins, increased sediment and chemical inputs (Euliss & Mushet 1996, Guntenspergen et al. 2002; DeKeyser et al. 2003), alterations to the upland and wetland soils (Euliss & Mushet 1996), and grazing, mowing, and burning the landscape (Guntenspergen et al. 2002; DeKeyser et al. 2003) have altered the quality of wetlands within the PPR (Euliss & Mushet 1996; Guntenspergen et al. 2002; DeKeyser et al. 2003). The resulting landscapes can have adverse effects on the function of the wetland even when the wetland basins are not directly disturbed (i.e. if disturbances occur on the surrounding land) (Guntenspergen et al. 2002; Gilbert et al. 2006).

In the Midwestern United States, more than 50 percent of depressional wetlands by area have been converted to agricultural production in the last century. Temporary and seasonal wetlands are especially susceptible to anthropogenic disturbance because, compared to semipermanent wetlands, these basins hold water for a shorter period of time (Detenbeck et al. 2002; DeKeyser et al. 2003; Mita et al. 2007). The periodic dry cycles, characteristic of the PPR, allow seasonal and temporary wetland basins to be cultivated during dry years reducing the quality and quantity of wetland habitat in the region (Detenbeck et al. 2002; van der Kemp et al. 2016).

The loss of wetland habitat has resulted in decreasing populations of waterfowl and increasing incidents of flooding along rivers and streams (Galatowitsch & van der Valk 1996). Since the early 1990s, the rate of wetland loss has slowed, but land use on adjacent land has intensified, continuing to threaten wetland habitats (Guntenspergen et al. 2002). In North Dakota, more than 50 percent of the land bordering wetland basins is in agricultural production, where very few of the remaining wetlands are undisturbed (Guntenspergen et al. 2002).

Wetland function and integrity are vulnerable to chemical, physical, and biological alterations to the surrounding landscapes (Guntenspergen et al. 2002). Urbanization and agricultural development promote wetland degradation as wetland hydrology is altered and these systems receive increased sediment and nutrient inputs (Euliss & Mushet 1996; Guntenspergen et al. 2002; Miller & Wardrop 2006; USEPA 2016), which can lead to eutrophication (Guntenspergen et al. 2002). In addition, increased sedimentation can increase water turbidity resulting in reduced productivity and germination rates of native wetland vegetation (Dentenbeck et al. 2002; Miller & Wardrop 2006), decreased plant species richness (Guntenspergen et al. 2002; Miller & Wardrop 2006), increased occurrence of non-native and weedy species (Miller &

Wardrop 2006), and decreased waterfowl populations (Detenbeck et al. 2002; Guntenspergen et al. 2002).

Managing lands for agriculture, such as draining wetland basins, alters the natural hydrology of the landscape, which can affect wetland basins that remain (Euliss & Mushet 1996). Wetlands surrounded by agricultural land with altered hydrology can increase the erosion potential of the landscape (Guntenspergen et al. 2002). When native vegetation is removed for cultivation, surface runoff is affected and can result in increased inputs of sediments and chemicals into the wetland basins (Euliss & Mushet 1996; Dentenbeck et al. 2002). Lands without vegetation are unable to retain and slow the runoff of surface water, further manipulating hydrology (Euliss & Mushet 1996). In addition, increased pressures from agriculture can alter the snowpack distribution in the uplands which can affect the quantity of water wetlands receive from upland snowmelt, affecting the infiltration capacity of the soils (Conly & van der Kamp 2001).

Tillage within and around wetland basins can affect the condition of wetlands within agricultural landscapes. The need to estimate the condition of wetlands is a result of increased stress on wetland resources due to agriculture and the correlation between wetland condition and wildlife populations (Cowardin et al. 1981). Wetland condition can be degraded as multiple external factors interact with the basins and surrounding landscapes. However, a lack of disturbance can also negatively affect wetland condition as the vegetation in the PPR for example, evolved under natural disturbances such as drought, flooding, grazing, and fire (DeKeyser et al. 2003; Euliss & Mushet 2011). To develop wetland management programs and policies for the PPR, it is valuable to monitor the long term condition and land use strategies of the region’s resource (Conly & van der Kamp 2001).

Wetland Assessment

The U.S. Fish and Wildlife Service (USFWS) has been documenting the extent of the nation's wetlands since the 1950s, however, comparatively few studies have documented wetland condition (USEPA 2016). Across the United States, wetland degradation has been a major environmental concern. Regulatory and conservation efforts have been developed to reduce wetland loss nationally (Guntenspergen et al. 2002). Legislation such as the Clean Water Act (1972), which focused on restoring and maintaining the integrity of the waters of the United States, and the Swampbuster provision in the Food Security Act (1985), which discouraged farmers from draining and filling wetlands, has resulted in an increased awareness of anthropogenic influences on the nation's wetlands (Carletti et al. 2004; Genet & Olsen 2008; Hargiss 2009; Gleason et al. 2011).

Historically, the focus of wetland assessment has been on the total wetland acreage to determine the extent of wetland resources to meet the national goal of "no net loss" (Dahl & Watmough 2007). Recording the extent of wetlands by monitoring location, size, type, and function has allowed for the documentation of wetland habitat loss. As wetland resources continue to be pressured by development, it is increasingly important to maintain updated data regarding the state of wetland resources (Dahl & Watmough 2007). Researchers began to develop methods to assess the function and quality of wetlands in the 1980s. Wetland assessment methods were designed to detect ecosystem stressors, document the effectiveness of current management practices, monitor wetland condition for regulatory purposes (Fennessy et al. 2007; Kentula 2007), and determine the impacts that human activities have on ecosystems and their biota (Stoddard et al. 2006).

Wetlands have variable physical, chemical, and biological characteristics due to climatic, geologic, and physiographic settings (Gilbert et al. 2006). The variability among wetlands across a landscape make it difficult to develop and implement scientifically defensible assessment methods that can be used on a large scale (Wardrop et al. 2007), therefore it is important to use methods that account for variability within and among wetlands (Gilbert et al. 2006). Reference sites, which are the least impacted sites and considered to be in the most natural condition, are often used to improve the scientific defensibility of the assessment and account for variations between wetlands (Brooks et al. 2004, Stoddard et al. 2006).

Designing assessment methods that account for wetland variability, detect modified functionality, and have limited time requirements has been a challenge (Gilbert et al. 2006). Collecting data from a representative sample to corroborate assessment methods can also be difficult because of challenges associated with gaining access to wetland sites (i.e. due to private land ownership) and the high cost often required to implement the assessments (Wardrop et al. 2007; Hargiss and DeKeyser 2014). Prior to sampling, determining the best method expected to yield the greatest amount of information with the available financial and time resources is important (Wardrop et al. 2007, Hargiss et al. 2017). Assessors should focus on methods that evaluate both indirect and direct environmental stressors that can affect variability among the associated biota (Stoddard et al. 2006).

Hydrogeomorphic Model

The Hydrogeomorphic (HGM) Model, developed by the United States Army Corps of Engineers and the Natural Resources Conservation Service, is a wetland assessment method that uses geomorphic setting, water source, and hydrodynamics to assess wetland function for better management of the resource (Gilbert et al. 2006; Hargiss et al. 2017). The HGM model has been adapted for different regions of the United States, including the PPR, to account for the variability among wetlands across the country (Guntenspergen et al. 2002; DeKeyser et al. 2003;

Gilbert et al. 2006; Wardrop et al. 2007; Hargiss 2009). The adapted HGM model for the PPR assesses wetland function with consideration of the region's landscape, soil, and vegetative characteristics (Gilbert et al. 2006; Hargiss et al. 2017).

Mathematical models calculate a functional capacity index (FCI) using the data collected from the HGM model for the wetlands assessed (Gilbert et al. 2006; Hargiss 2009). The FCI scores range from 0 to 1.0, where a wetland with a score of 1.0 has the assessed function characteristic of reference standard wetlands. Six functions are assessed with this model including 1) water storage; 2) groundwater recharge; 3) ability to remove particulates from the water column; 4) the ability to remove, convert, and sequester dissolved substances that enter the wetland; 5) the ability to support native plant communities with consideration of natural disturbance regimes; and 6) ability to provide faunal habitat (Gilbert et al. 2006).

Floristic Quality Index

The Floristic Quality Index (FQI) assesses the vascular plant species within a wetland and determines condition based on the quality of the vegetation (Gilbert et al. 2006; Hargiss et al. 2017). The quality of wetland habitat is determined using ecological conservatism and species richness of native plant community measurements (Miller & Wardrop 2006; USEPA 2016). The coefficient of conservatism (C value) expressing ecological conservatism is assigned to plant species based on the species specificity to particular environments and their sensitivity to disturbance. C values range from 0 to 10, where species with narrow geographic ranges and low tolerance for disturbance are given higher scores than those with more broad distribution and higher tolerance for disturbance. Non-native species have a C value of 0 (Miller & Wardrop 2006; USEPA 2016) or are not assigned a C value at all and thus are not used in calculating the FQI (NGPFQAP 2001). Due to the reliance on typical species distribution and species richness, the FQI has been found to be an adequate predictor of wetland condition (Miller & Wardrop 2006), and have been used to assess wetland condition across the United States (Hargiss et al. 2017).

Index of Biological Integrity

The Index of Biological Integrity (IBI) assesses the condition of an ecosystem based on biological communities known to be sensitive to environmental alterations (Karr 1981). A site with high biological integrity is expected to support a biological community with species composition, diversity, and function comparable to similar undisturbed habitats within the region (Mack 2007). Metrics are used to determine the biological integrity of a community of organisms sensitive to a gradient of anthropic disturbances (Mack 2006; Euliss & Mushet 2011). Using results from the metrics, condition categories are determined, which can aid in resources management (Euliss & Muschet 2011).

The Index of Plant Community Integrity (IPCI) was an IBI developed for wetlands in North Dakota. Plant communities are sensitive to anthropic disturbances (i.e. alterations to nature hydrology, cultivation, etc.) and were therefore determined to be the most appropriate measure of biological integrity (DeKeyser et al. 2003; Hargiss et al. 2008). Vascular plants are relatively large in size and easy to sample, provide valuable ecosystem functions, and have well-understood taxonomy at state and regional levels, making them useful biological communities in IBI assessments. IBI assessments using vascular plants can be a cost-effective way to sample wetland condition due to the well-established vegetation sampling methods and the ease of collecting vegetation data in the field (Mack 2007).

The IPCI for North Dakota analyzes nine metrics related to species richness, the number of native perennials, and the percentage of introduced and annual species within a wetland to determine the quality of the plant community (DeKeyser et al 2003; Hargiss et al. 2008). Wetland zones are sampled based on zone boundaries at the time of sampling. Therefore, data collected using IPCI methods documents the condition of the wetland at a specific point in time (Euliss & Mushet 2011).

Landscape Assessment Methods

Landscape assessment methods use remote sensing satellite imagery to monitor ecosystems across a region. Using this method, spatial and temporal data can be analyzed in a timely and cost-effective manner (Mita et al. 2007). Landscape assessments yield an estimation of wetland condition for a particular area. This assessment method is useful for the evaluation of condition across a set of watersheds for planning management projects (Brooks et al. 2004).

The Landscape Wetland Condition Assessment Model (LWCAM) overlays NWI maps and land use data to determine the condition of wetlands in an area. This model separates wetlands into categories of good, intermediate, and poor based on the percentage of grassland, the largest patch of grassland, and the number of patches of each type of land use within a 300 meter buffer surrounding the wetland (Mita et al. 2007; Hargiss 2009). Landscape models monitor natural (i.e. succession), and anthropic changes (i.e. agricultural production, drainage) in wetland condition (Dahl & Watmough 2007).

Rapid Assessment Methods

Rapid assessment methods (RAM) classify wetland condition and identify possible stressors to the biotic community using observable field indicators. These assessment strategies required less time and financial resources than other sampling methods (Carletti et al. 2004; Fennessy et al. 2004; Hargiss 2009, Stein et al. 2009; Hargiss et al. 2017). RAMs use defined metrics with categories based on the assumption that the condition of wetlands will vary along a disturbance gradient (Stein et al. 2009). RAMs are useful in documenting anthropogenic impacts on wetland systems and can be used to evaluate management practices and restoration and mitigation projects (Fennessy et al. 2004; Wardrop et al. 2007). RAMs require less time and financial resources in the field, while also reducing the required taxonomic expertise of the surveyor, which allows for increased sample sizes at lower costs (Fennessey et al. 2007). RAMs are valuable assessment methods when in-depth assessments are too time-consuming and expensive to complete (Hargiss et al. 2017).

Multi Assessment Approach

To increase confidence in the data set, multiple assessments methods can be used to determine the condition of a wetland. The United States Environmental Protection Agency (USEPA) recommends using a three-tier approach where wetlands are assessed using three methods that vary in effort, scale, quality of data, and future use of results to address a variety of wetland monitoring objectives (Kentula 2007). Each level of assessment yields information regarding wetland condition at different levels of detail. Methods utilizing multiple types of assessments have been developed across the United States and are often adapted to regional and local wetland landscapes (Hargiss et al. 2017). A combination of previously discussed methods is used to assess the condition of the wetland using the multi-tier approach.

The first tier of the assessment is the landscape method. The condition of the wetland is determined using landscape indicators that have been shown to correlate with field condition to give an estimate of the average wetland condition in a particular area. The landscape method can be completed with low-cost requirements as the work can be completed in an office on the computer. This first level of assessment can aid in determining areas that should be addressed for future management (Kentula 2007).

The second level of the three-tier approach is the rapid assessment. These assessments have low time and field effort requirements to determine the condition of the wetland. Rapid assessments give an approximation of the condition based on hydrology, plant communities, and soils within the wetland and can be used to assess the impacts of human activities (Kentula 2007). These assessments can be used to identify the environmental stressors affecting the biological communities within the assessed wetland; however, RAMs cannot yield information about ecological interactions (Hargiss et al. 2017).

The third level involves an intensive assessment of biological, chemical, and/or morphological data, which will be analyzed to assess the condition of the wetland (Kentula 2007). The intensive assessments require a significant amount of effort and collection of a large amount of data for a comprehensive analysis resulting in the corresponding condition of the wetland (Kentula 2007; Hargiss 2009). The HGM model and FQI are commonly used as level three assessments (Hargiss et al. 2017). The third level of assessment results in detailed data regarding the condition and function of the assessed wetland, however, these assessments are expensive and time consuming (Hargiss et al. 2017).

The three-tiered method can help validate the results from each level of assessment, increasing the confidence in the condition evaluation (Kentula 2007; Hargiss 2009). Data collected from multi-tiered assessments of wetlands can be utilized by landowners, managers, and government agencies to determine the condition of wetlands as well as the stressors affecting wetlands. This information can be used to improve management strategies within these ecosystems (Hargiss et al. 2017). Trends in wetland condition can be identified through repeat assessments of wetlands in an area, and allow for the development of long-term data (Cressey et al. 2016; Hargiss et al. 2017).

State Assessments

In California, the California Rapid Assessment Method (CRAM) is used to monitor the condition of the wetlands statewide (USEPA 2016). The CRAM was developed to monitor wetlands in California with a standardized protocol. This method relies on the assumption that wetland structure is correlated with ecosystem function. Wetland condition is based on four characteristics including 1) buffer and landscape context; 2) hydrology; 3) physical structure; and 4) biotic structure. Each category is given a score which is compiled into an overall condition score, where higher scores indicate better condition (USEPA 2016). Using the CRAM, the data indicate estuarine wetlands in California are in better condition compared to depressional wetlands. Estuarine wetlands are exposed to fewer direct stressors because they are generally part of a larger network of connected salt marshes compared to the depressional wetlands which are smaller and often fragmented (USEPA 2016).

In Wisconsin, the Floristic Quality Assessment (FQA) uses species richness and the coefficient of conservation (C value) to calculate the FQI for wetlands across the state (USEPA 2016). Researchers found sites with increased phosphorus and nitrogen inputs had higher plant productivity, which artificially inflated FQI scores. A weighted C value was used to account for the artificially high FQI scores and standardize the data (USEPA 2016). Using this assessment, the data show palustrine emergent wetlands were dominated by invasive species such as reed canary (*Phalaris arundinacea)* and hybrid cattail (*Typha x glauca*) compared to the other wetland community types. This information was intended to improve the management of wetland resources in Wisconsin (USEPA 2016).

In Nebraska, a statewide assessment utilized the multi-tiered approach to determine the condition of the wetlands in the state’s priority landscapes (USEPA 2016). Researchers grouped sites into land use categories (i.e. Wildlife Management Areas (WMA), Waterfowl Production Areas (WPA), privately owned) for the level one assessment. The Nebraska Wetland Rapid Assessment Method (NeW_RAM) was developed to use as a level two assessment. The level three assessment involved using the FQI and surveying the amphibian communities which helped develop a protocol to assess and monitor the amphibian populations in relation to wetland management. Within all landscape regions across the state, FQI scores were found to be linked to land use. In areas of the state where all the wetlands sampled had FQI scores below reference scores, it is hypothesized that sites near reference condition no longer exist because of land use changes and natural and anthropic disturbances. This statewide condition assessment of wetlands in Nebraska can be used to improve future conservation and management efforts in the state (USEPA 2016).

Regional Assessments

Regional assessments are valuable because natural ecosystems are fluid and not confined to constructed boundaries of individual states, therefore wetlands within a region may share ecological characteristics (USEPA 2016). The HGM Model and IBI assessment methods are valuable measures of wetland function and condition when assessing a single site, however, on a regional scale, these methods are often too intensive for practical use (Guntenspergen et al. 2002). The USEPA’s Environmental Monitoring and Assessment Program (EMAP) developed a landscape model to assess wetland condition within various geographic ranges in the United States (Rosen et al. 1995). This assessment method evaluates biological integrity, productivity, water quality, and flood prevention. In this case, biological integrity on a regional scale utilizes reference site characteristics to determine natural variations in wetland condition including vegetation patterns and hydrologic cycles (Rosen et al. 1995).

Regional assessments were completed as part of the USEPA's National Wetland Condition Assessment (NWCA) where data was compiled based on four identified ecoregions of the United States (Figure 1.2) (USEPA 2016). The Vegetation Multimetric Index (VMMI) was developed to determine the biological condition of wetlands in the four ecoregions of the United States. The VMMI uses four metrics 1) FQI; 2) relative importance of native species; 3) number of plant species tolerant to disturbance; and 4) relative cover of native monocot species. The West ecoregion was found to have 21 percent of the wetlands sampled in good condition, 18 percent in fair condition, and 61 percent in poor condition. The Coastal Plains ecoregion had 50 percent of the wetlands in good condition, 21 percent in fair condition, and 29 percent in poor condition. The Eastern Mountains and Upper Midwest ecoregion had 52 percent in good condition, 11 percent in fair condition, and 37 percent in poor condition. The Interior Plains ecoregion had 44 percent in good condition, 36 percent in fair condition, and 19 percent in poor condition (USEPA 2016).

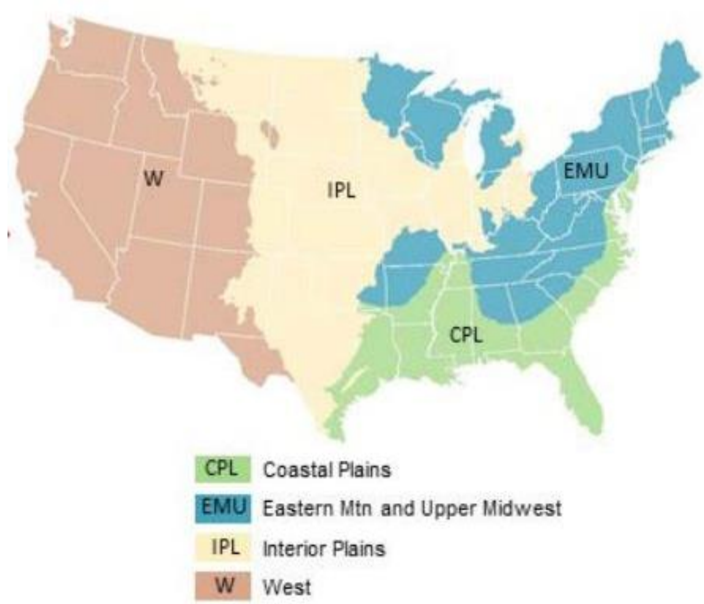

Figure 1.2. Map of the ecoregions of the United States used by the USEPA for the NWCA (Map from USEPA 2016).

North Carolina, South Carolina, Georgia, and Alabama conducted a regional assessment of forested wetlands. This regional assessment measured wetland condition using 1) vegetation mean C (i.e. metric based on the Coefficient of Conservation for plant species); 2) invasive species cover; 3) Amphibian Quality Assessment Index (AQAI); 4) macroinvertebrate diversity; 5) Buffer Landscape Development Intensity Index (LDI); 6) Ohio Rapid Assessment Method; 7) water quality nutrients; and 8) soil metal information (USEPA 2016). This assessment found that areas with dense populations and intensive agriculture had poorer condition wetlands than areas with less disturbance. Studies like these can improve both state and regional management of wetland resources to improve ecosystem services (USEPA 2016).

Within the PPR, upland landscapes surrounding wetlands can influence the condition of the basin. Guntenspergen et al. (2002) suggested completing wetland assessments at both the landscape and basin scales to gain a better understanding of the stressors on wetlands based on land use gradients. Landscape indicators can include spatial density, land use characteristics of both the uplands and the wetlands, measurements of seasonal surface water loss and wetland drainage, and habitat potential through estimations of dabbling duck breeding pairs (Guntenspergen et al. 2002). The method involves prioritizing wetland function, determining and evaluating potential wetland indicators to distinguish between wetland conditions, and differentiate between the variability of each indicator (Guntenspergen et al. 2002). Within the PPR, there was a correlation between wetland condition and the cropland to upland ratio. However, it is recommended that the method be recalibrated yearly to improve accuracy since hydrology has a defining effect on wetland habitats in the PPR (Guntenspergen et al. 2002).

National Assessments

Nationally, wetland extent has been documented by the USFWS since the 1950s, however, wetland condition has not been well documented (USPEA 2016). To manage wetland ecosystems, it is important to understand the condition of wetland resources across the United States. As part of the National Aquatic Resource Surveys (NARS) the USEPA designed the National Wetland Condition Assessment (NWCA) to document the condition of the nation's wetlands to improve public policy regarding wetland resources (USEPA 2016).

The NWCA uses standardized protocols across the nation to collect scientifically defensible data yielding valuable information regarding the condition of wetlands across the United States (USEPA 2016). The first NWCA was completed in 2011 and the second in 2016. The goal is to assess the nation's wetlands every five years to document the change in national wetland condition over time (USEPA 2016).

The sample population for the NWCA includes tidal and nontidal wetlands within the continental United States with open water less than 1 meter deep, with rooted vegetation, and not in crop production at the time of sampling (USEPA 2016). Sample sites are selected using USFWS S&T plots to fit the population. In the first NWCA in 2011, 48 percent of the wetlands were found in good condition, 20 percent in fair condition, and 32 percent in poor condition across the country. This data will be used to compare wetland condition overtime in the continental United States (USEPA 2016).

Wetland Condition in North Dakota

Wetlands in the Northern and Northwestern glaciated plains of North Dakota were used to develop the IPCI wetland assessment, which resulted in the development of wetland categories that reflected very good, good, fair, poor, and very poor wetland condition (DeKeyser et al. 2003). Wetlands in very good and good condition were found in relatively intact landscapes surrounded by native prairies. Wetlands in fair condition had significantly more disturbances within the catchment compared to wetlands in higher condition categories. The disturbances at fair condition wetlands likely degraded the plant community by introducing and improving the competitive advantages of invasive plant species, resulting in a reduced condition of the wetlands. Wetlands found to be in poor and very poor condition were in highly disturbed areas where anthropic activities resulted in exotic plant species accounting for a high percentage of the plant community (DeKeyser et al. 2003).

A study of 255 seasonal wetlands in the Missouri Coteau ecoregion utilized samples from multiple assessment methods and a probabilistic sampling design to account for the regional condition of wetlands (Hargiss 2009). The LWCAM found 44 percent of the wetlands in good condition, 4 percent in intermediate condition, and 52 percent in poor condition. The IPCI found 18 percent in very good, 18 percent in good condition, 16 percent in fair condition, 20 percent in poor condition, and 27 percent in very poor condition. The NDRAM found 38 percent of the wetlands in good condition, 12 percent in fair high condition, 35 percent in fair low condition, and 15 percent in poor condition. Topography appeared to determine land use and therefore condition, where flat areas were more likely to be disturbed for agricultural production and areas with more topography were more likely to be grazed and often more native (Hargiss 2009).

Within the PPR, wetlands were sampled using the IPCI where anthropic disturbances remained constant throughout a four year study (Euliss & Mushet 2011). Although the disturbance regime remained constant, IPCI scores varied for individual wetlands between years, where over 60 percent of the wetlands sampled moved between condition categories over the course of the study. Variations in IPCI scores are the result of the natural fluctuations in wetland systems often a result of the hydrological dynamics. Semipermanent wetlands had IPCI scores ranging from 45 to 95 (condition ranging from fair to good) where condition score varied an average of 19.3 units for individual wetlands between years. Seasonal wetlands had similar results, where wetlands had IPCI scores ranging from 34 to 91 and individual wetland scores varied by an average of 21.1 units between years (Euliss & Mushet 2011). These types of assessments illustrate the dynamic nature of prairie wetland systems. Species composition responds to the natural fluctuating hydrologic regime, which affects IPCI scores based on species composition. During wet cycles, IPCI scores are likely to be higher compared to IPCI scores during dry cycles. These assessments are snapshots in time, so wetlands should be assessed annually to accurately document shifts in wetland condition and to develop more robust data sets (Euliss & Mushet 2011).

Introduction

The Prairie Pothole Region (PPR) accounts for approximately 274,500 square kilometers of the Northern Great Plains in the United States. This region is characterized by high concentrations of depressional wetlands, known as prairie potholes (Guntenspergen et al. 2002; Niemuth et al. 2010; Gleason et al. 2011; Dahl 2014). Prior to European settlement approximately two million hectares of wetlands spotted the landscape of North Dakota. However, due to anthropic activities particularly land use conversions from native grasslands to agricultural fields (Cowardin et al. 1981, Galatowitsch & van der Valk 1996; Dahl 2014), less than half of those hectares remained by 1984 (Leitch & Baltezore 1992; Dahl 2014). The loss of wetlands has slowed since the early 1900s, but land use has intensified on adjacent lands and continues to threaten wetlands in the PPR (Guntenspergen et al. 2002).

Wetlands in the PPR are predominately temporary, seasonal, and semipermanent (Gleason et al. 2011; Dahl 2014), with seasonal wetlands accounting for the greatest number of wetlands in the region (Stewart & Kantrud 1971, Hargiss 2009). In North Dakota temporary and seasonal wetlands account for 93 percent of the total number of wetlands, but they only account for 43 percent of the acreage of water in the state (DeKeyser et al. 2003).

Understanding the condition of the wetlands is important to providing a complete representation of the status of wetlands across the country (Genet & Olsen 2008). As more land is impacted by anthropic activity, it is important to document anthropogenic land use alterations and its impact on wetland systems. The United States Environmental Protection Agency (USEPA), the North Dakota Department of Health, and North Dakota State University collaborated to assess the condition of wetlands throughout the state every five years. The first assessment was completed in 2011, and the second assessment was completed in 2016. Monitoring current conditions of wetlands in North Dakota is an important first step in improving wetland quality and habitat. This study aims to document the condition of wetlands in North Dakota and analyze how the condition changed between 2011 and 2016 using the North Dakota Rapid Assessment Method (NDRAM).

Methods

Site Selection

We assessed 44 wetlands in 2011 and 39 wetlands in 2016 as part of the USEPA's National Wetland Condition Assessment (NWCA) (Figure 1.3). We sampled wetlands selected from a list of randomized points across North Dakota generated by the USEPA, which used their protocol for the NWCA site selection (USEPA 2011). The USEPA used Status and Trends (S&T) plots developed by the U.S. Fish and Wildlife Service (USFWS) and Generalized Random Tessellation Stratified (GRTS) design to generate sites representative of the nation's wetlands (USEPA 2011).

The target population of the NWCA included all wetlands in the conterminous US with rooted vegetation and open water less than one meter deep. Cropped wetlands were included in the target population if the wetland was not in crop production at the time of sampling (USEPA 2016). We eliminated points that did not fit the target population. Following point selection, we contacted landowners to gain permission to survey the wetlands. Based on where we were able to gain permission, 20 percent of the wetlands assessed in this study were on public lands, although public lands account for less than 10 percent of the land area in North Dakota (Gleason et al. 2011).

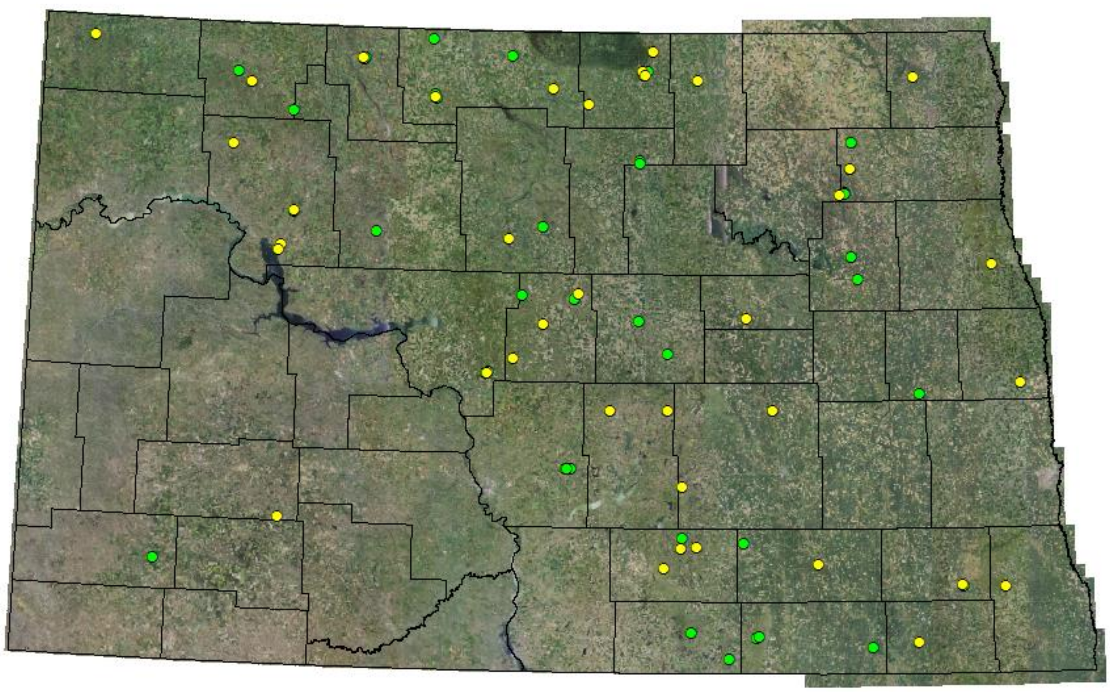

Figure 1.3. Map of Sites Sampled. Sites in green were sampled in 2011 and sites in yellow were sampled in 2016.

Sampling & Analysis

We sampled wetlands in the summers of 2011 and 2016 using the North Dakota Rapid Assessment Method (NDRAM) as developed by Hargiss (2009) (see Appendix A). The survey included: photos taken in each cardinal direction from the point; a determination of the hydrologic classification; basic description of the site; and documentation of land use and/or disturbances within the wetland and the surrounding area. The NDRAM requires the assessor to answer a series of questions regarding possible stressors to wetlands within the PPR, organized into three subcategories or metrics – surrounding land use, hydrology and habitat alterations, and vegetation. A set of answers with denoted numerical values accompany each question. Following the NDRAM protocol, each metric was assigned numerical values as follows: Metric 1 (M1) - surrounding land use 20 points; Metric 2 (M2) - hydrology and habitat alterations 57 points, and Metric 3 (M3) - vegetation 23 points. We then summed the metric values to yield a final total score between 0 and 100 and compared the total score to predetermined condition categories. The condition categories are good (total scores of 69-100), fair high (total scores of 53-68), fair low (total scores of 27-52), and poor (total scores of 0-26).

Although the target population of the NWCA included all wetlands in the conterminous US with rooted vegetation and open water less than one meter deep, we chose to focus this study on temporary, seasonal, and semipermanent wetlands located in North Dakota because the majority of wetlands within the PPR are within these hydrologic classes. In addition, the NDRAM was specifically developed to determine the condition of typical prairie pothole wetlands (i.e., temporary, seasonal, and semipermanent wetlands) (Hargiss 2009) rather than the broad wetland types included in the NWCA target population. While other wetlands types exist within the PPR of North Dakota, it is valuable to concentrate on the temporary, seasonal, and semipermanent wetlands to gain the best understanding of the current condition of prairie pothole wetlands.

We used t-tests to determine whether there were differences in average NDRAM scores between the wetlands assessed in 2011 and 2016 in North Dakota, regardless of hydrologic class. To this end, we used t-tests to compare the total scores and individual metric scores in all wetlands assessed in 2011 or 2016 (i.e., all 44 temporary, seasonal, and semipermanent wetlands sampled in 2011 vs. all 39 temporary, seasonal, and semipermanent wetlands sampled in 2016). In addition, we used t-tests to examine how wetlands in different hydrologic classes fared across North Dakota by comparing the NDRAM scores sampled in 2011 and 2016 by hydrologic class (i.e., the 20 seasonal wetlands sampled in 2011 vs. the 25 seasonal wetlands sampled in 2016).

Results

In 2011, nine of the wetlands assessed were in good condition, 15 were in fair high condition, 13 in fair low condition, and seven wetlands were in poor condition. In 2016, eight wetlands were in good condition, five in fair high condition, 23 in fair low condition, and three in poor condition (Figure 1.4). The mean total condition score for prairie pothole wetlands (across all hydrologic classes) was 53.23 (SE= 3.44) (fair high) in 2011 and 49.46 (SE= 3.07) (fair low) in 2016. The overall wetland condition scores did not change significantly between 2011 and 2016 ($p=0.42$) when scores were compared across hydrologic classes (Figure 1.5).

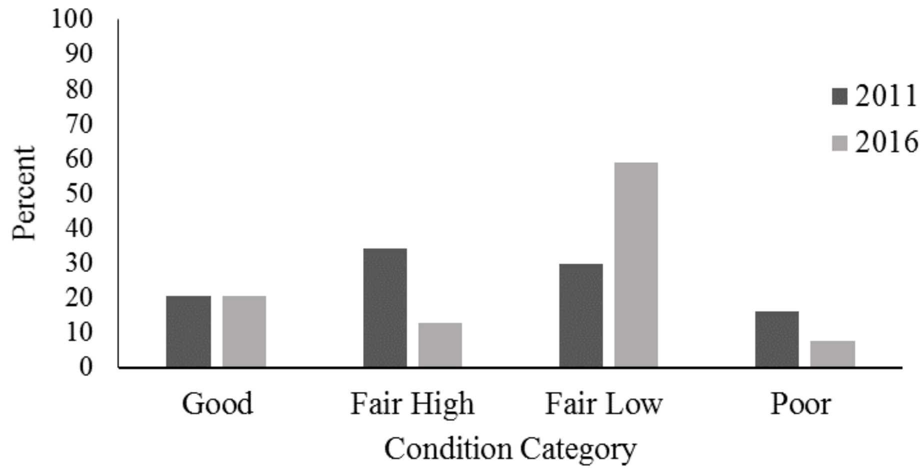

Figure 1.4. Percent of prairie pothole wetlands per condition category in 2011 (n=44) and 2016 (n=39). A higher percentage of wetlands were found in fair low condition in 2016, with a lower percentage of wetlands in fair high and poor condition compared to 2011.

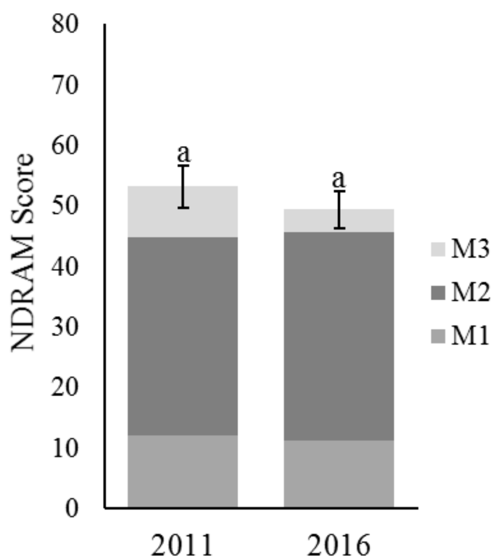

Figure 1.5. Total NDRAM scores in 2011 and 2016. The average total score in 2011 was 53.23 (n= 44) and in 2016 the average total score was 49.43 (n= 39). No significant difference was found in total score between 2011 and 2016. (M1= Metric 1; M2= Metric 2; M3= Metric 3).

In 2011, the mean metric scores for all hydrologic classes were 12.20 (SE= 1.05) for surrounding land use (M1), 32.59 (SE= 1.78) for hydrology and habitat alterations (M2) and 8.43 (SE= 0.88) for vegetation (M3). In 2016, the mean metric scores for M1, M2 and M3 were 11.28 (SE= 0.89), 34.51 (SE= 1.70) and 3.66 (SE= 0.92), respectively. There were no differences in

M1 (p= 0.51) and M2 (p= 0.44) when the individual metrics for 2011 and 2016 were compared across hydrologic classes using t-tests. In contrast, the mean M3 score (8.43 (SE= 0.88)) was greater in 2011 than in 2016 (3.66 (SE= 0.92)) (p= 0.0003).

When hydrologic classes were considered separately, one temporary wetland was found to be in good condition in 2011, one was in fair high condition, three in fair low condition, and two in poor condition (n= 7). In 2016, one temporary wetland was in good condition, one in fair high condition, one in fair low condition, and zero in poor condition (n= 3) (Figure 1.6). Temporary wetlands had a mean total score of 41.86 (SE= 11.02) (fair low) in 2011 and 60.33 (SE= 7.69) (fair high) in 2016. However, when the NDRAM total scores for temporary wetlands sampled in 2011 were compared to temporary wetlands sampled in 2016 using t-tests, there was no difference between the 2011 and 2016 overall mean scores (p= 0.34) (Figure 1.7).

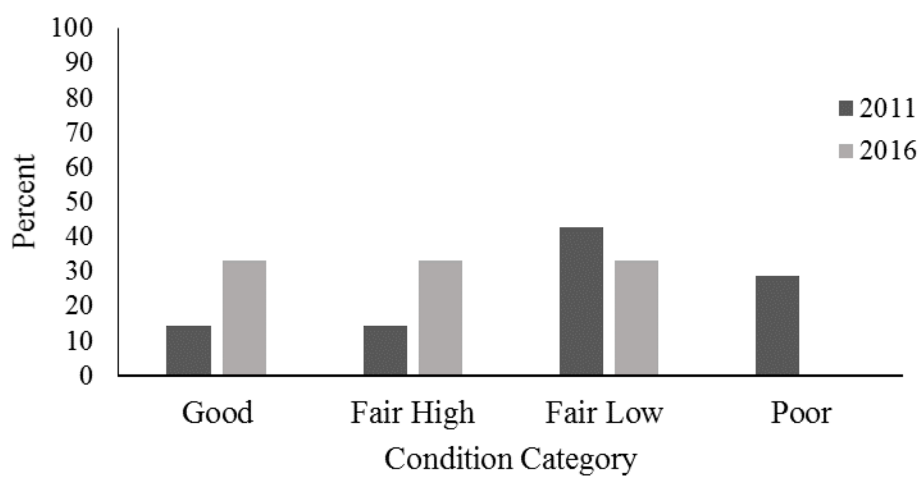

Figure 1.6. Percent of temporary wetlands per condition category. A higher percentage of temporary wetlands sampled in 2016 (n= 3) were in good and fair high condition compared to 2011 (n= 7).

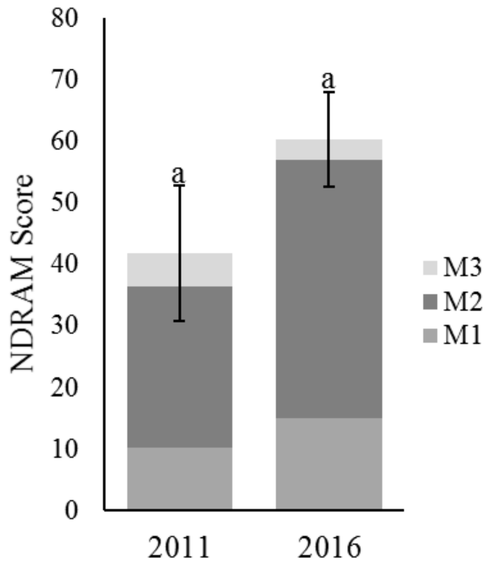

Figure 1.7. NDRAM scores in temporary wetlands. The average score of temporary wetlands in 2011 was 41.86 (n= 7) and 60.33 in 2016 (n= 3). (M1= Metric 1; M2= Metric 2; M3= Metric 3).

In 2011, four seasonal wetlands were found to be in good condition, six were in fair high condition, five in fair low condition, and five in poor condition (n= 20). In 2016, seven seasonal wetlands were in good condition, two were in fair high condition, thirteen in fair low condition, and three in poor condition (n= 25) (Figure 1.8). Seasonal wetlands had a mean total score of 48.90 (SE= 5.32) (fair low) in 2011 and 50.00 (SE= 4.47) (fair low) in 2016. When the NDRAM total scores for seasonal wetlands sampled in 2011 were compared to seasonal wetlands sampled in 2016 using t-tests, there were no significant differences between the overall mean scores in 2011 and 2016 (p= 0.87) (Figure 1.9).

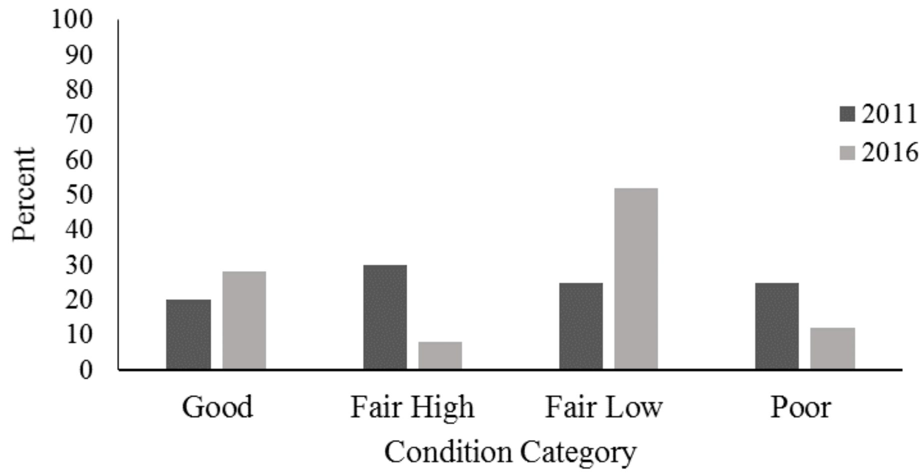

Figure 1.8. Percent of seasonal wetlands per condition category. A higher percentage of wetlands were found to be in fair low condition in 2016 (n= 25) compared to 2011 (n= 20). A higher percentage of wetlands sampled in 2011 were in fair high and poor condition compared to 2016.

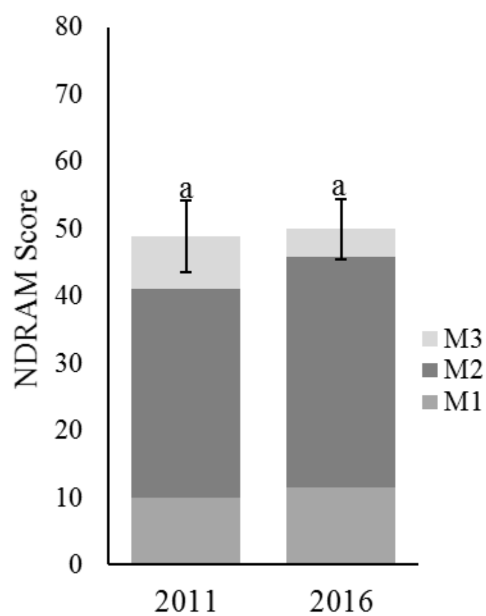

Figure 1.9. NDRAM scores in seasonal wetlands. The average score of seasonal wetlands in 2011 was 48.90 (n= 20) and 50.00 in 2016 (n= 25). (M1= Metric 1; M2= Metric 2; M3= Metric 3).

In 2011, four semipermanent wetlands were found to be in good condition, eight were in fair high condition, five were in fair low condition, and zero in poor condition (n= 17). In 2016, zero semipermanent wetlands were in good condition, two were in fair high condition, nine in fair low condition, and zero in poor condition (n= 11) (Figure 1.10). Semipermanent wetlands had a mean total NDRAM score of 63.00 (SE= 3.68) (fair high) in 2011 and 45.27 (SE= 3.15) (fair low) in 2016. When the total NDRAM score for semipermanent wetlands sampled in 2011 were compared to semipermanent wetlands sampled in 2016 using t-tests, there was significant difference between the overall mean scores (p= 0.002) (Figure 1.11).

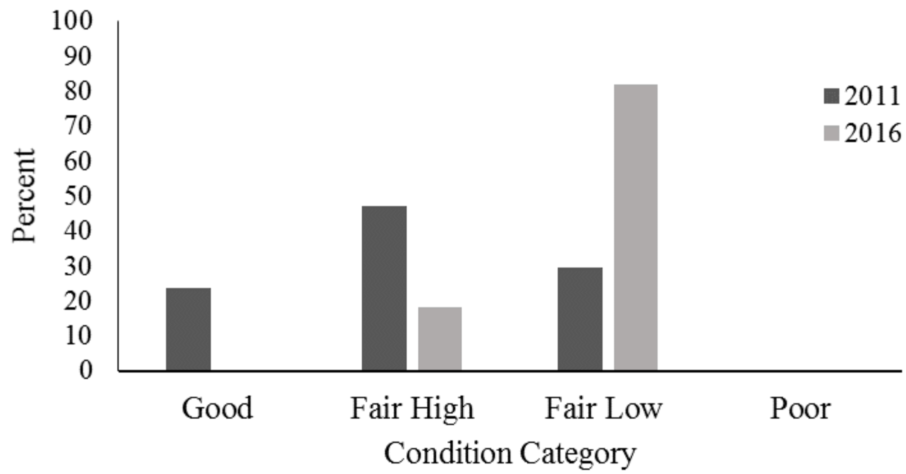

Figure 1.10. Percent of semipermanent wetlands per condition category. A higher percentage of semipermanent wetlands were found to be in fair low condition in 2016 (n= 11) compared to 2011 (n= 17).

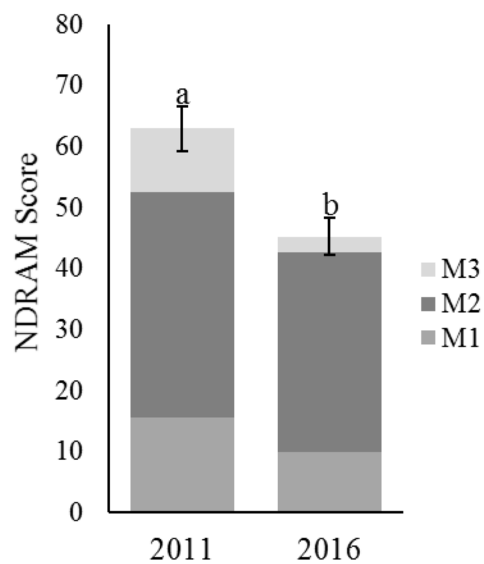

Figure 1.11. NDRAM scores in semipermanent wetlands. The average score of semipermanent wetlands in 2011 was 63.00 (n= 17) and 45.27 in 2016 (n= 11). (M1= Metric 1; M2= Metric 2; M3= Metric 3).

Discussion

In 2011 and 2016, the majority of the sampled wetlands were in fair condition (i.e. fair high or fair low), with more wetlands in fair condition in 2016. Although there was no detectable significant difference in overall wetland condition based on NDRAM scores between 2011 and 2016, it does appear that wetland condition may be decreasing. The greatest decrease in NDRAM metric score between 2011 and 2016 was Metric 3 (M3) which determines condition based on the presence of native and invasive species (Hargiss 2009; Hargiss et al. 2017). This may be the result of continued conversions of grasslands for agricultural production. North and South Dakota have been shown to have the highest concentration of hectares converted from native grasslands to agricultural lands east of the Missouri River in a study of North and South Dakota, Minnesota, Nebraska, and Iowa (Wright & Wimberly 2013). The loss of native grassland could result in increased opportunities for non-native vegetation to move into wetland areas and begin to dominate the plant communities.

The overall NDRAM score for temporary wetlands was higher in 2016, but the increased score is likely a result of sample size. In 2011, seven temporary wetlands were sampled compared to three sampled in 2016. Within temporary wetlands, Metric 2 (M2) showed the greatest variation between 2011 and 2016. M2 measures wetland condition based on disturbances including soil, vegetation, and habitat alteration (Hargiss 2009; Hargiss et al. 2017). The increase in M2 scores is most likely an artifact of the sample population versus of an indication of changes in disturbance regimes in North Dakota. The cyclic wet and dry periods, characteristic of the PPR, commonly result in the cultivation of temporary wetlands during dry years (Dentenbeck et al. 2002; van der Kamp et al. 2016). North Dakota experienced more precipitation in 2011 compared to 2016, so in 2016 it is likely that more temporary basins were cultivated, which could have excluded some poor condition wetlands from the study.

Seasonal wetlands had no significant changes in overall condition between 2011 and 2016. In 2016, there was an increase in the percentage of seasonal wetlands found to be in fair low condition compared to 2011. In addition, in 2016 there was a decrease in the percentage of wetlands found in both fair high and poor condition, while there was an increase in the percentage of wetlands found in good condition. This could suggest a number of possibilities including 1) seasonal wetlands classified as fair high condition in 2011 have decreased in condition to fair low, 2) seasonal wetlands classified as poor condition may have improved condition categories to fair low, and 3) seasonal wetlands classified as fair high in 2011 have been improved and are now in good condition. Since the overall condition of seasonal wetlands in North Dakota remained relatively unchanged between 2011 and 2016, the changes in each condition category may be a result of natural fluctuations in wetlands due to the dynamic hydrology within prairie wetlands, where shifts in condition class can occur even when disturbances regimes remain unchanged (Euliss & Mushet 2011).

In 2016, the overall condition of semipermanent wetlands was significantly lower than the semipermanent wetlands sampled in 2011. No wetlands sampled were found to be in poor condition in either 2011 or 2016. However, in 2011, approximately 24 percent of the semipermanent wetlands sampled were in good condition and in 2016 none of the semipermanent wetlands sampled were in this condition category. Semipermanent wetlands in fair low condition increased between 2011 and 2016. Interestingly, comparing individual metric scores, M3 decreased the most between the two years. This decrease in M3 scores (i.e. the native vegetation metric) could suggest that the cover of invasive species is increasing and/or the diversity of the plant community is decreasing in semipermanent wetlands in North Dakota. Continuing to monitor the condition of semipermanent wetlands will allow for a more complete assessment in the condition trends for these wetlands. However, the decreased condition observed between 2011 and 2016 will be important to continue monitoring because semipermanent wetlands account for a majority of the wetlands in the PPR by hectare (Stewart & Kantrud 1971; DeKeyser et al. 2003).

The main goal of wetland condition assessments is to detect stressors on wetland resources and determine the quality of wetland function in the landscape (Stoddard et al. 2006; Fennessy et al. 2007; Kentula 2007). For this study, the NDRAM was used to quickly assess the condition of prairie pothole wetlands in North Dakota. RAMs are a cost-effective assessment method that are valuable when assessing the effects of anthropic activities on wetland basins (Fennessy et al. 2004; Wardrop et al. 2007; Hargiss 2009; Stein et al. 2009; Hargiss et al. 2017). It is important to understand the limitation of RAMs, although these assessments are commonly validated against other wetland assessment methods. The NDRAM was corroborated with assessment methods known to accurately evaluate wetland condition in North Dakota including the Index of Plant Community Integrity (IPCI), the Hydrogeomorphic (HGM) Model, the Floristic Quality Index (FQI), and the Landscape Wetland Condition Assessment Model (LWCAM) (Hargiss 2009; Hargiss et al. 2017). However, RAMs cannot provide detailed data on wetland function and species health and therefore should not be used to replace more intensive assessments methods (Stein et al. 2009). On a statewide scale, RAMs are valuable to sample a large number of wetlands during the growing season (Fennessy et al. 2004; Wardrop et al. 2007; Hargiss 2009; Stein et al. 2009; Hargiss et al. 2017).

All wetland condition assessments yield information regarding the condition of wetlands at a particular point in time (Euliss & Mushet 2011). Therefore, it is most valuable to repeat assessments to determine patterns and trends in the condition data (Hargiss et al. 2017). Continuing to monitor wetland condition is important, especially in the PPR, because of the agricultural potential and the vulnerability to anthropic disturbances (Gilbert et al. 2006; Gleason et al. 2011). Although seasonal and temporary wetlands are most susceptible to land use conversions to agriculture from natural areas (Dentenbeck et al. 2002; DeKeyser et al. 2003; Mita et al. 2007), these wetlands showed no significant difference in overall condition between sample years. To fully document patterns in wetland resource condition in North Dakota, continuing to monitor wetland condition will be vital.

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CHAPTER 2: AN ANALYSIS OF SOIL AND VEGETATIVE PROPERTIES IN A FEN IN EDDY COUNTY, NORTH DAKOTA

Abstract

Fens are rare wetland types where groundwater inputs determine water levels and chemistry which impact the soil and vegetative properties. Water levels in fens remain relatively constant throughout the growing season resulting in saturated soils and providing habitat to rare vegetation. Within the Prairie Pothole Region (PPR) of North Dakota, fens are rare and not well studied. This study seeks to characterize the soil and vegetative properties of a fen in Eddy County, North Dakota.

We sampled five vegetative communities that appeared to be dominated by distinct vegetation. Three 10 meter square vegetation plots were selected for sampling within each of the five vegetative communities. Vegetation data was analyzed for species richness, evenness, diversity, and the percent of introduced species. We described the soil profile and collected soil samples from a soil pit within two meters of each vegetation plot. Soil samples were tested for percent organic matter, pH, and electrical conductivity of the surface horizon.

Multivariate analysis determined all five vegetative communities were significantly different. Univariate analysis determined a significant difference in species richness between the herbaceous and cattail communities and the tree and cattail communities. There were also significant differences in evenness and diversity between the herbaceous and cattail communities. Cattail communities also had significantly more introduced species compared to the other four vegetative communities. The soil data indicated no significant difference in percent organic matter, pH, or electrical conductivity between the five vegetative communities.

The data from this study yields information regarding species diversity and soil properties found in a natural fen and can be used to aid in fen restoration and conservation projects.

Literature Review

Fens are one of the rarest and most biologically diverse wetland ecosystems in the temperate regions of the United States (Amon et al. 2002; Bedford & Godwin 2003; van Diggelen et al. 2006; Jassey et al. 2014), and account for only a small portion of the overall landscape (Bedford & Godwin 2003). These wetland systems are commonly found in regions previously glaciated; developing in areas where groundwater discharge results in extensive areas of saturation that extend from the plant root zones to the soil surface (Bedford & Godwin 2003). The interaction between ground and surface water determines the function of these systems (Amon et al. 2002; Bedford & Godwin 2003) and impacts the soil and vegetative properties (Bedford & Godwin 2003).

Classifications

Fens are classified as groundwater fed peatlands with pH values of 4.2 or greater near the surface. Most fens are non-tidal palustrine emergent wetlands with saturated organic soils, graminoid vegetation, and pH values ranging from 5.5 to 7.4 (Cowardin et al. 1979; Almendingr & Leete 1998a). Fens are locally rare but geographically widely dispersed because of the particular soil and water properties that characterize these systems (Jimenez-Alfaro et al. 2012). In fens, peat accumulation is derived from *Carex* species (Amon et al. 2002), with water and nutrient inputs from the groundwater supply (Heller & Zeitz 2012), compared to bogs which have peat derived from *Sphagnum* mosses (Amon et al. 2002) with water and nutrients from precipitation (Heller & Zeitz 2012).

Fens can be classified as poor or rich fens. Poor fens are nutrient poor and often slightly acidic. Rich fens are nutrient rich due to mineral accumulation, often calcium accumulations, in the groundwater, which results in pH values between six and eight (Jassey et al. 2014). Fens classified as prairie rich fens are found in open landscapes and formed in glacial lake plains and drainage ways. Prairie fens are dominated by graminoid vegetation and often lack shrub and tree vegetation (Aaseng et al. 2005). Fens classified as calcareous are peatlands with spring-seepage zones and are dominated by vegetation tolerant of high concentrations of calcium carbonates (Almendinger & Leete 1998a).

Ecosystem Services

Fens are important systems for water retention, drought prevention, nutrient removal, and carbon sequestration. Saturated conditions slow decomposition, allowing carbon to be stored (Heller & Zeitz 2012). In addition to the carbon storing potential, fens have become a priority for conservation because these systems provide habitat to a large number of endangered and threatened species (Jassey et al. 2014; Fernandez-Pascual et al. 2015).

Calcareous fens provide habitat to calciphitic vegetation. In Minnesota, calciphitic flora found in calcareous fens account for a number of the state-threatened plant species (Almendinger & Leete 1998a). Calcareous fens are critical habitats in prairie ecosystems due to the valuable habitat provided to rare species and the high diversity found at these sites (Almendinger & Leete 1998a).

Geology

In North American peatlands, climate and geology influence ecological condition and can have a significant impact on the soil and vegetative properties (Nekola 2004). Geologic deposits surrounding fens influence soil permeability and mineralogy due to groundwater movement. Fens are often found on landscapes with permeable coarse-grained deposits, which allow vertical hydraulic gradients to form (Almendinger & Leete 1998a). Hydrologic gradients form in areas with stratigraphic and/or topographic breaks which force groundwater to the surface (Amon et al.

2002). Infiltrating groundwater will move laterally as it reaches a less permeable layer, and will discharge at the surface where that layer reaches the hillslope. The accumulation of organic matter will generally occur at slope breaks where groundwater discharge is great enough to promote peat formation (Amon et al. 2002).

Hydrology

Groundwater is the main determinant of water chemistry in fens, which differentiates this wetland type from other prairie pothole wetlands, where surface water determines water chemistry (Bedford & Godwin 2003). The water and soil in these systems are often base-rich (Bedford & Godwin 2003) with dissolved minerals (Turner et al. 2000) and depleted of nitrogen (N) and phosphorus (P) (Bedford & Godwin 2003; Turner et al. 2000), due to the movement of groundwater as it passes through and around bedrock, glacial deposits, and soil (Bedford & Godwin et al. 2003).

Groundwater affects the soil properties and vegetation composition in fens. Water levels remain relatively stable throughout the growing season due to the groundwater inputs (Amon et al. 2002; Bedford & Godwin 2003), however, changes in water levels have been found to be the most important hydrologic factors (Malmer 1986; Amon et al. 2002; Nekola 2004). Fluctuations in water levels have been found to influence soil properties such as pH, redox potential, and decomposition (Malmer 1986), as well as determine vegetation patterns across a single site (Malmer 1986; Nekola 2004). For example, Malmer (1986) found differences in bog vegetation at a regional scale could be correlated to the fluctuations in hydrology at the site.

High evapotranspiration rates combined with low groundwater inputs can result in greater seasonal variation in water levels. Fens can have reduced groundwater inputs from the natural state with anthropic or climatic alterations which can lower groundwater levels. Reduced groundwater inputs can increase decomposition and reduced the accumulation of organic matter (Amon et al. 2002).

Within peatlands, nutrient availability is dependent on the origin of the water inputs (i.e. bogs are dependent on nutrients that enter the system through precipitation, whereas fens rely on the movement of groundwater through bedrock and sediments) (Vitt & Chee 1990). Calcium, magnesium, and sodium are abundant in the surface water of both bogs and fens. However, nitrogen and phosphorus are commonly limited, which affects the primary productivity of the peatland. Water chemistry does impact the vegetation found within peatlands, however, fluctuations in water levels is thought to have a larger role in plant community development (Vitt & Chee 1990).

Vegetation

Vegetation also differentiates fens from other peatlands. Bogs do not have characteristic species but generally will lack the flora considered to be fen specialists. Poor fens are dominated by *Sphagunum* and rich fens are dominated by bryophytes and vascular plants (Malmer 1986). The variation in vegetation between peatland types is thought to be controlled by acidity and alkalinity, nutrient availability, and the depth to the groundwater (Tousignant et al. 2010; Jassey et al. 2014). However, factors such as the thickness of the peat layer and shading also contribute to plant community composition (Tousignant et al. 2010).

Fens have many rare species and high levels of biodiversity (Amon et al. 2002; Kolli et al. 2009; Jimenez-Alfaro et al. 2012). These wetlands generally lack tree and shrub communities, but if trees and shrubs are present they are often stunted and isolated developing along the margins of the fen (Malmer 1986; Aaseng et al. 2005). In a study of a calcareous fen in Estonia, 13 rare species were found across a 7,552 ha mire complex (Ilomets et al. 2010). The high species diversity and specialized fen vegetation may be the result of adaptations to low nutrient conditions (Almendinger & Leete 1998b; Bedford & Godwin 2003). Under the saturated conditions characteristic of fens, plant material does not decay, so N and P are immobilized in the organic layer and are not available as nutrients for growing plants. Some fen specialist vegetation has adapted to the nutrient poor conditions in areas of groundwater discharge, which would otherwise be nutrient limiting (Turner et al. 2000). Among the highly specialized plant species wetland generalist species are also prevalent (Amon et al. 2002). Fen vegetation is generally dominated by bryophytes, sedges, herbs, and grasses (Bedford & Godwin 2003).

Understanding vegetation patterns gives insight into proper management and restoration practices. In Estonia variations in plant species composition was a result of seasonal fluctuations in water level, water pH, and conductivity (Ilomets et al. 2010). In Iowa, variation in vegetation was dependent on pH, Ca, and Mg (Nekola 2004). Variations in water levels along with stability and duration of low water levels in the summer and soil organic matter have also been found to determine vegetation patterns in these ecosystems (Ilomets et al. 2010), where water levels are most often correlated to vegetation gradients within fens (Nekola 2004).

The highest diversity and rare species counts were found on low vegetation mats in Iowa fens (Nekola 2004). In New Zealand, plant communities dominated by *Sphagnum* were found in areas with lower and more stable groundwater level compared to areas dominated by *Carex* and *Baumea* species (Sorrell et al. 2007). The communities dominated by *Carex* and *Baumea* species experienced fluctuating hydrology with both wet and dry periods. Wetter areas within the fen were found to be dominated by *Carex* species, whereas drier sites were dominated by forbs and graminoid species (Sorrell et al. 2007). In Minnesota, prairie rich fens are dominated by graminoid vegetation and have a moss cover of less than 25 percent (Aaseng et al. 2005). These fens may have patchy shrub layers less than two meters tall and consist of willow species (Aaseng et al. 2005).

In southeastern Quebec, Canada, five different plant communities were determined within a single peatland complex (Tousignant et al. 2010). The communities included wooded fens, disturbed wooded fens, shrubby fens, highly disturbed fen, and highly disturbed bog. In this study, the five communities were found to have varying characteristics. The wooded fens were found to be in areas with thick peat deposition and significant tree cover. Disturbed wooded fens had higher water pH and conductivity values. The shrubby fens were found in wetter areas with higher groundwater levels and thin peat deposition. The shrubby fens also had lower tree cover and more disturbance. The highly disturbed fens and bogs were found closer to the margins of the study site and near drainage ditches. These areas had low tree cover and the water table experienced greater fluctuations. In these highly disturbed areas, the indicator species present were non-peatland species. Overall, this study found that groundwater depth and tree cover were the most important factors determining the composition of the vegetation (Tousignant et al. 2010).

In western Poland, vascular plants were found to respond to concentrations of iron in the soil. Deep-rooted vegetation did not show significant difference in species richness and diversity along the poor to rich gradient. However, shallow rooted vegetation was found to have reduced richness and diversity within poor fens compared to rich fens (Jassey et al. 2014). The greatest species richness and diversity were found in what was classified as moderately rich and rich fens. Jassey et al. (2014) inferred that the results indicate that optimal conditions for diversity exist with moderate pH, calcium accumulations, and depth to groundwater supply.

Soils

Fen soils are commonly saturated with water at the soil surface during the growing season (Amon et al. 2002; Bedford & Godwin 2003). These soils are often classified as calcareous histosols with soil profiles reflecting the local variability of the soil material and hydrology (Slaughter 1999). Saturated conditions maintain anoxia, resulting in slow decomposition and the accumulation of organic matter (Amon et al. 2002). Organic soils form under waterlogged conditions and will have 20 percent organic matter or greater within 30 centimeter of the surface (Davis & Lucas 1959; Mitsch & Gooselink 2015). The water storage potential and rate of water movement through the soil are dependent on the porosity and structure of the soil, with high rates of decomposition decreasing the size of organic particles creating smaller pores and increasing the bulk density (Boelter 1969).

Water chemistry and vegetation can determine the properties of organic soils (Davis & Lucas 1959; Walter et al. 2016). Organic soils can be eutrophic, oligotrophic, or mesotrophic. Eutrophic organic soils are found in areas that have groundwater with high mineral content and often support tree and shrub vegetation. Oligotrophic organic soils are found in areas where groundwater has low nutrient content, which inhibits the growth of many types of vegetation but will promote the growth of mosses and rushes. Mesotrophic organic soils will have moderate concentrations of minerals (i.e. an intermediate between eutrophic and oligotrophic soils). (Davis & Lucas 1959).

The formation processes and the associated plant communities in bogs and fens result in organic soils with different characteristics. Bog soils are associated with acid tolerant vegetation, and have water that is acidic with low concentrations of nutrients (i.e. oligotrophic). In comparison, fen soils are formed through the decomposition of cattails, rushes, sedges, and grasses producing fibrous soils, with high calcium concentrations (Davis & Lucas 1959).

Interactions between surface and groundwater, geochemistry, geology, climate, plant community composition, and land use impact soil development and chemistry (Guntenspergen et al. 2002; Heller & Zeitz 2012). Within and among fens there are variations in the degree of peat decomposition. For example, in areas with comparable rates of decomposition, landscapes with peat accumulation derived from *Sphagnum* (i.e. bogs) will have higher organic carbon content compared to peat formed by vascular plant species such as *Carex* (i.e. fens) (Walter et al. 2016). Fens are commonly dominated by fibric peat, which is characterized as having low decomposition, however sapric peat, highly decomposed plant material, is found in fens with lower water inputs or increased drainage (Amon et al. 2002), as the decomposition of peat increases with soil aeration (Walter et al. 2016).

Degradation

Fen degradation is a result of habitat deterioration, fragmentation, and climate change (Klimkowska et al. 2010), along with the drainage of surrounding land for agriculture, and the use of groundwater for irrigation (van Diggelen et al. 2006). These changes impact the flora utilizing fen habitats. Alteration to fen hydrology is thought to be the main cause of degradation of the vegetative communities within these ecosystems. Changes to the groundwater source or recharge conditions can alter mineral loading and nutrient availably within the entire system (Almendinger & Leete 1998b).

Anthropic activity such as hydrologic alterations, pollution of both ground and surface water due to agricultural practices, and increased sulfur and nitrogen inputs, can result in eutrophication, acidification, and desiccation, further degrading fen ecosystems (van Diggelen et al. 2015). Fens in the temperate zones worldwide are well suited for agriculture (i.e. calcareous nature, nutrient rich, easy to drain), making these systems particularly vulnerable to anthropogenic disturbances (Jablonska et al. 2011). Eutrophic fens have lower species diversity as fen specialist species are out competed by other vegetation (van Diggelen et al. 2015). Increased rainwater inputs result in acidification as peat accumulation increases. This process is accelerated when *Sphagnum* becomes the dominant vegetation (van Diggelen et al. 2015). In an analysis of peatlands in southeastern Quebec, Canada, anthropic disturbances resulted in reduced species richness. Peatland bryophytes and vascular plants had reduced richness and exotic species were favored. However, this study maintained that abiotic factors were the main controllers of plant community composition at larger peatland complex scales (Tousignant et al. 2010).

Anthropogenic disturbance can result in decreased water levels and increased sedimentation (Gleason et al. 2011); which can result in reduced productivity (Almendinger & Leete 1998b; Gleason et al. 2011) and monotypic plant communities (Gleason et al. 2011). Understanding the complex water dynamics within these wetland systems, can aid in the maintenance of biodiversity and productive vegetative systems (Gleason et al. 2011). Sedimentation and increased nutrient inputs can reduce species diversity as introduced species become established and outcompete native vegetation. In a study of the Cheboygan Marsh in Michigan, areas dominated by *Typha x glauca* had decreased species diversity as *T. glauca* outcompeted the native vegetation (Angeloni et al. 2006).

Fens can be degraded even by small scale drainage. Following drainage events, shrubs and trees often move into these areas due to the changes in hydrology (Jablonska et al. 2011). In western Europe, fen degradation has resulted from draining these systems for agricultural production. Lowering the groundwater level has reduced soil fertility in this region (Zeitz & Velty 2002), as aeration degrades these soils (Schindler et al. 2003). Peatlands store carbon which can be released by anthropic activities often associated with agricultural use. Land use changes can promote decomposition and mineralization which releases the stored carbon into the atmosphere. These activities can drastically alter the physical, chemical, and biological properties of peat soils as peat formation processes are manipulated often resulting in subsidence (Heller & Zeitz 2012).

When fens are flooded and there is increased water flow through the fen, there will be increased inorganic nutrient inputs (Malmer 1986). In New Zealand, flooding was found to impact soil properties such as porosity, structure, conductivity and oxidation, which altered the distribution of plant communities. Flooding can change plant community composition as flood tolerant species replace flood intolerant species as oxygen availability decreases with increasing water levels (Sorrell et al. 2007).

The type of disturbance along with its timing, extent, and duration has the ability to impact and change the plant community within a fen (Amon et al. 2002). Anthropic activities such as tilling, ditching, filling, and draining fens manipulate the depth to the groundwater supply and impact the moisture at the root zone. These alterations change the hydrology of the system and can reduce the groundwater inputs thus increasing the influence of surface water. Runoff from precipitation can remove the minerals from the system and increase oxygen in the surface horizons (Amon et al. 2002).

Succession

Fen hydrology promotes specialized vegetation well-adapted for nutrient poor conditions, creating habitats for endangered and rare plants. In natural fens, succession is suppressed by the hydrologic conditions (i.e. saturation) (Jablonska et al 2011). Plant species composition can be altered if *Sphagnum* becomes a dominant species resulting in the formation of floating mats and the increased thickness of the peat layer. This shift increases the influence of rainwater and decreases the influence of surface and groundwater. A hydrologic shift occurs as base-poor rainwater replaces base-rich groundwater, shifting the vegetative community from that of a rich fen to a poor fen, which can eventually become a woodland habitat (van Diggelen et al. 2015).

Fire and grazing aid in maintaining open fens dominated by herbaceous vegetation. Without these management tools, fens are often invaded by woody species and become more shrub dominated. Bart et al. (2016) found that plowed and natural fens were vulnerable to the encroachment of woody species. However, regardless of land use, the invasion of woody plants was patchy across all fens.

Restoration

The main goal in fen restorations is to restore the natural hydrology and nutrient supplies (Sorrell et al. 2007). The best way to restore the hydrology is to reestablish the relationship between the hydrology and the top soil, where base-rich surface and groundwater can infiltrate the soil (van Diggelen et al. 2015). In a restoration of a New Zealand fen, reestablishing the groundwater levels was found to aid in the management of non-native fen vegetation (Sorrell et al. 2007). Increasing the groundwater levels allowed for flood-tolerant vegetation to compete with the flood-intolerant species that had moved into the site. Reflooding may not fully restore the plant community if the organic matter and nutrients are depleted from soil oxidation when the area is drained. The elimination of the organic matter and soil nutrients can result in subsidence, which can hinder attempts to restore hydrology through reflooding (Sorrell et al. 2007).

Simply rewetting the landscape and allowing soil saturation to occur is often not sufficient to restore fen ecosystems (Sorrell et al. 2007). Natural fens may have seasonality allowing for the appropriate conditions for fen specialist vegetation. Failure to restore the groundwater seasonality and cyclic wet and dry periods can inhibit the restoration potential of the plant community. Once the hydrology is restored, upland vegetation that became established during the drainage period may remain competitive inhibiting the establishment of fen plant communities. The restoration of fen vegetation may also be inhibited by an insufficient seed bank on the restoration site or disturbances by fauna that occupy the landscape limiting the development of the desired plant community (Sorrell et al. 2007).

Introduction

Fens are rare wetlands that rely on groundwater inputs to provide minerals and nutrients to the plant community (Turner et al. 2000; Amon et al. 2002; Bedford & Godwin 2003; van Diggelen et al. 2006). The groundwater influence on these systems results in saturated soils which maintains anoxic conditions and slows rates of decomposition (Amon et al. 2002). Fens provide habitat to many rare plant species and commonly have high diversity due to the saturated and nutrient poor conditions (Almendinger & Leete 1998b; Turner et al. 2000; Amon et al. 2002; Bedford & Godwin 2003; Kolli et al. 2009; Jimenez-Alfaro et al. 2012; Fernandex-Pascual et al. 2015).

Understanding the soil and vegetative properties within fens can aid in the restoration and conservation of fens across the region. Fens are vulnerable to habitat deterioration and fragmentation along with climate change (van Diggelen et al. 2006; Klimkowska et al. 2010). Fens are particularly vulnerable to environmental alterations due to the dependence of fen vegetation on the groundwater supply for the nutrient conditions (Almendinger & Leete 1998b; Fernandex-Pascual et al. 2015).

Very little research has been done to describe the soil and vegetative properties in fens within the Prairie Pothole Region (PPR). Stewart and Kantrud (1972) described the dominant plant species in fens with emergent vegetation and Slaughter (1999) analyzed the vegetative and soil properties of a fen in North Dakota to infer the process of fen development. This study seeks to describe the soil and vegetative properties of a fen in central North Dakota and determine possible properties that result in the formation of certain vegetative communities across a single site.

Methods

Study Site

Our study site is in the Drift Plains ecoregion of Eddy County, North Dakota (Bluemle 1965) (Figure 2.1) (47.726899N; -98.663795W). The landforms throughout the Drift Plains were formed by glacial activity during the late Wisconsinan glaciation (25,000 and 20,000 years ago), characterized by flat to gently rolling hills (Bluemle 1965). In addition, glacial activity during the late Wisconsinan resulted in high concentrations of temporary and seasonal wetlands (Bryce et al. 1998). The vegetation throughout the region is characterized as the zone between the tallgrass and shortgrass prairies (i.e. mixed grass prairie) (Bluemle 1965; Bryce et al. 1998).

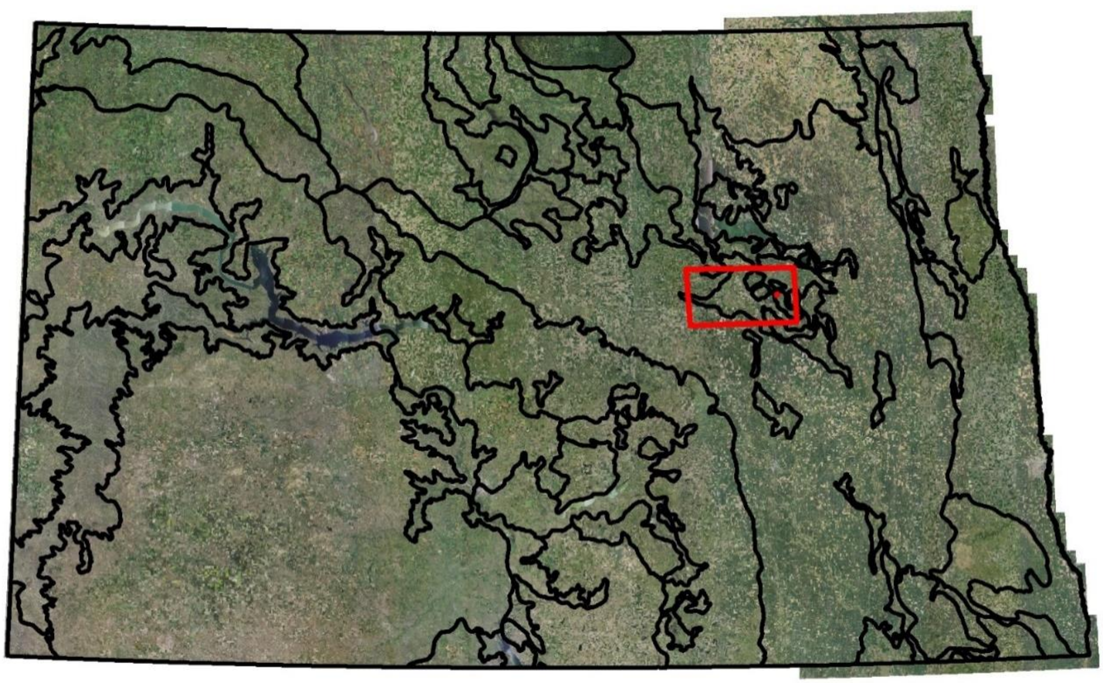

Figure 2.1. Map of North Dakota showing the fen study site (red circle) within Eddy County (red outline). Omernik Ecoregions (as produced by North Dakota Game and Fish Department) are outlined in black.

Vegetation and Soil Sampling

In 2016, we determined boundaries for five distinct plant communities by surveying visible changes in the dominant plant species (Figure 2.2). The resultant wetland plant communities were denoted cattail (*Typha* spp.), floating mat, herbaceous, tree, or wet meadow. We then selected three sampling locations within each wetland plant community that were representative of the overall plant community (Figure 2.3) resulting in 15 plots. We sampled the vegetation, examined the soil profile, measured the height of hummocks, and measured the depth to water during the 2017 growing season.

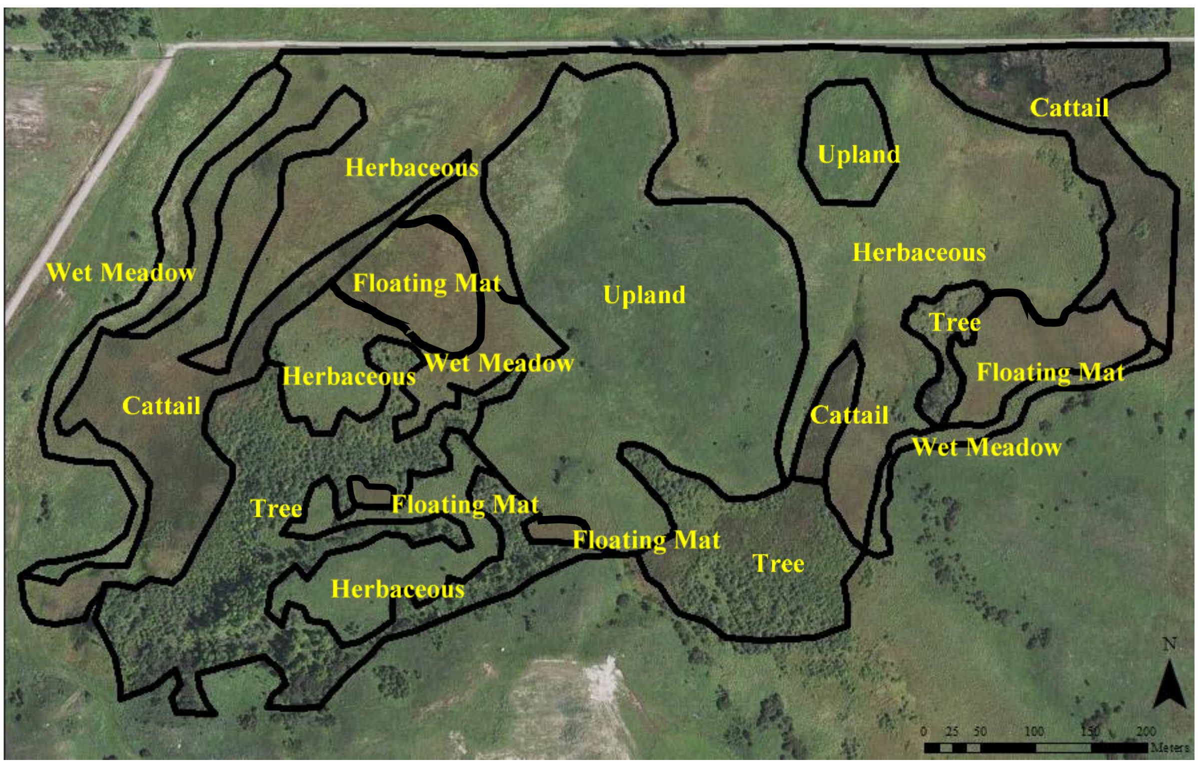

Figure 2.2. Map of distinct plant communities within the study area used to select sample locations within the five wetland plant communities (Cattail, Floating Mat, Herbaceous, Tree, Wet Meadow).

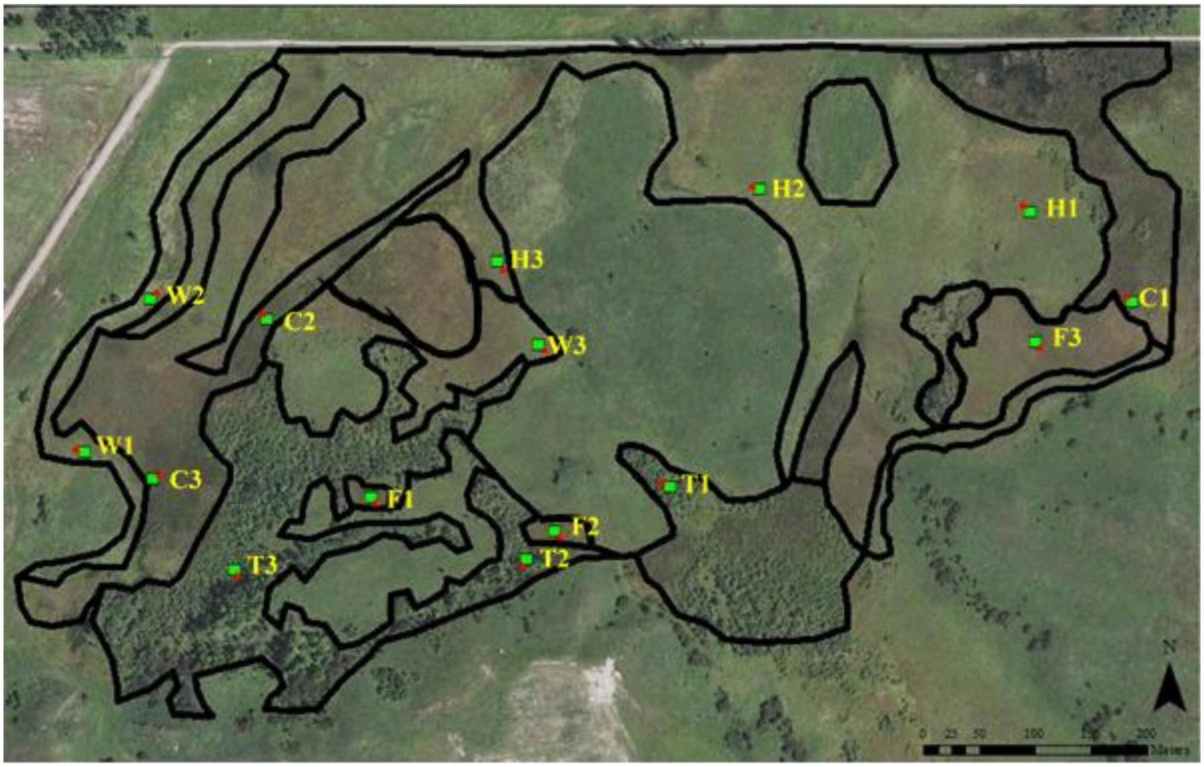

Figure 2.3. Sampling locations within wetland plant communities. Vegetation plots are indicated by green squares. Soil plots are indicated by red triangles. (C= Cattail; F= Floating Mat; H= Herbaceous; T= Tree; W= Wet Meadow).

We sampled the vegetation using 10 m x 10 m plots in July of 2017. Within each plot, we estimated the percent cover of each species encountered and calculated the relative cover of each species to determine species richness (S), evenness (E), diversity, and percent of introduced species (% I) within each plot. We calculated E using ln(S), to yield a value between zero and one. Higher E values indicate more diverse and more even plots. We used the Simpson Diversity Index (D) because the indices focus on the species the plots have in common versus the rare species in the plots. We calculated D using $1- (\sum n(n-1))/(N(N-1))$, where n is the relative cover of each species, and N is the total relative cover of each plot. D values range from zero to one, where values of zero indicate low diversity, and values of one indicate high diversity.

We sampled the associated soils within two meters of a corner of the vegetation plot with the most representative vegetation of the wetland plant community. We used an auger to collect soil to 100 cm and described the profile by documenting the lower boundary of each horizon, texture, wet color, structure, prevalence of roots, and porosity following the *Field Book for Describing and Sampling Soils* (Scheneberger et al. 2012). We compiled an average profile for each horizon using the average depth to horizon breaks and reported the texture, wet color, structure, prevalence of roots, and porosity based on the most common features among the three plots per vegetative community. Additionally, samples from each horizon were analyzed by the North Dakota State University Soil Testing Laboratory to obtain percent organic matter, pH, and electrical conductivity.

We measured the hummocks present in one square meter areas within the most representative corner of the vegetation plot and within one square meter surrounding the soil pit. We calculated the average hummock height per plot by compiling the data from the vegetation plot and the soil pit. Then, we calculated the average hummock height per community type. In addition, we measured the depth to water at the time of soil sampling (June 2017) per plot and calculated the average depth to water by vegetative community.

Analysis

We used both multivariate and univariate techniques (in particular, nonmetric multidimensional scaling (NMS), multi-response permutation procedure (MRPP), one-way analysis of variance (ANOVA), and Tukey's honestly significant difference (HSD) test) to thoroughly examine and compare the vegetative composition of our plant communities following Kobiela et al. (2017).

Prior to multivariate analysis, we transformed the relative cover data using the arcsine square root transformation (McCune and Grace 2002; McCune and Mefford 2011). To examine the composition of plant communities, we used NMS (relative Sørenson distance measure) and MRPP. We used NMS to depict the relationships among our vegetative communities and to determine the correlation between individual species and the ordination axes. We used MRPP on the relative cover of all species found at our fen to make comparisons between vegetative communities (we adjusted P-values using the Benjamini-Hochberg correction to account for multiple comparisons (Quinn & Keough 2002)).

We also analyzed our vegetation dataset using ANOVA to compare species richness (S), evenness (E), diversity (Simpson's; D), and the average relative cover of introduced species (% I) among the wetland plant communities (cattail, floating mat, herbaceous, tree, and wet meadow) encountered at our study site. Similarly, we used ANOVA to compare percent organic matter (% OM), pH, and electrical conductivity (EC) of the surface horizon encountered at each soil sampling location. We then used Tukey's (HSD) test to make pairwise comparisons among our wetland plant community types for both the vegetation and soils data.

Results

Vegetation

We surveyed a total of 150 plant species throughout the five wetland plant communities (see Appendix B). Our NMS analysis of the relative cover of all species at our site produced a final solution with three dimensions (final stress=5.31, instability= 0.00; Figures 2.4-2.6). We examined the Pearson correlation (r) of individual species with each NMS axis (Appendix C). We determined there is a negative correlation between Axis 1 and the species typically found in the cattail and floating mat communities, Axis 2 and the species typically found in the floating mat communities, and Axis 3 and the species typically found in the tree communities.

Our MRPP analysis of the relative cover data determined that the species composition of each vegetative community was distinct, i.e. each plant community type was significantly different than all other plant communities (Table 2.1). Cattail communities were significantly different from floating mat ($p= 0.035$), herbaceous ($p= 0.031$), tree ($p= 0.031$), and wet meadow (p= 0.031) communities. Floating mat communities were significantly different from herbaceous (p= 0.031), tree (p= 0.031), and wet meadow (p=0.031) communities. The herbaceous communities were significantly different from the tree (p=0.031) and wet meadow (p=0.038) communities. The tree communities were significantly different from the wet meadow communities (p=0.031).

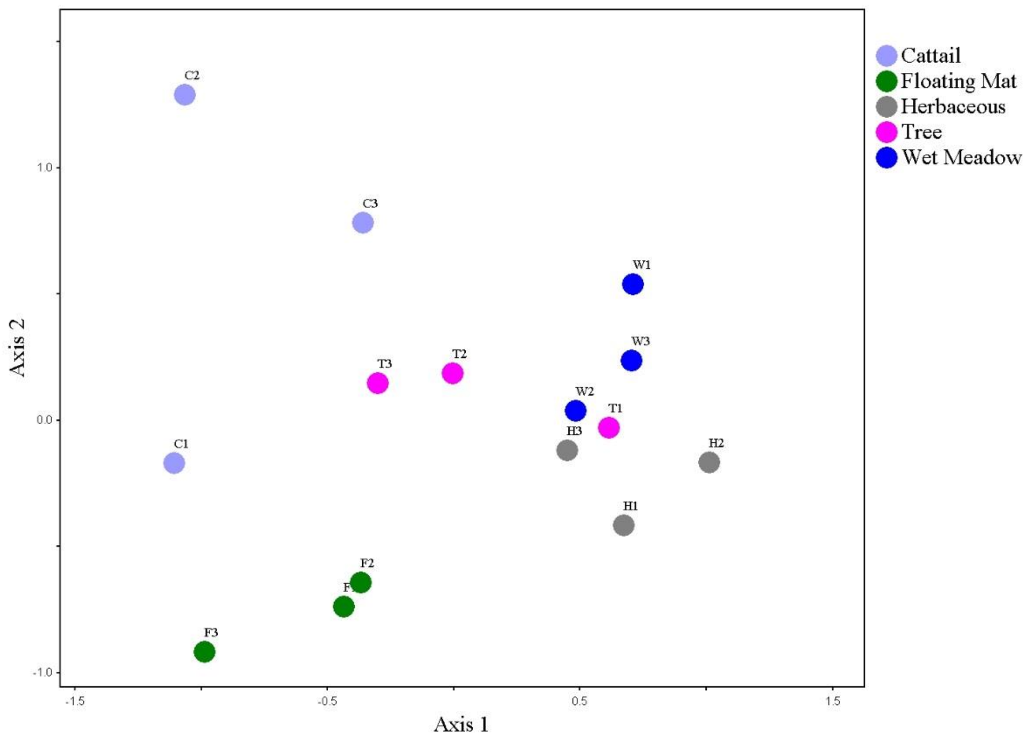

Figure 2.4. Nonmetric multidimensional scaling ordination of relative cover by vegetative community, Axis 1 versus Axis 2. Each symbol represents a single site.

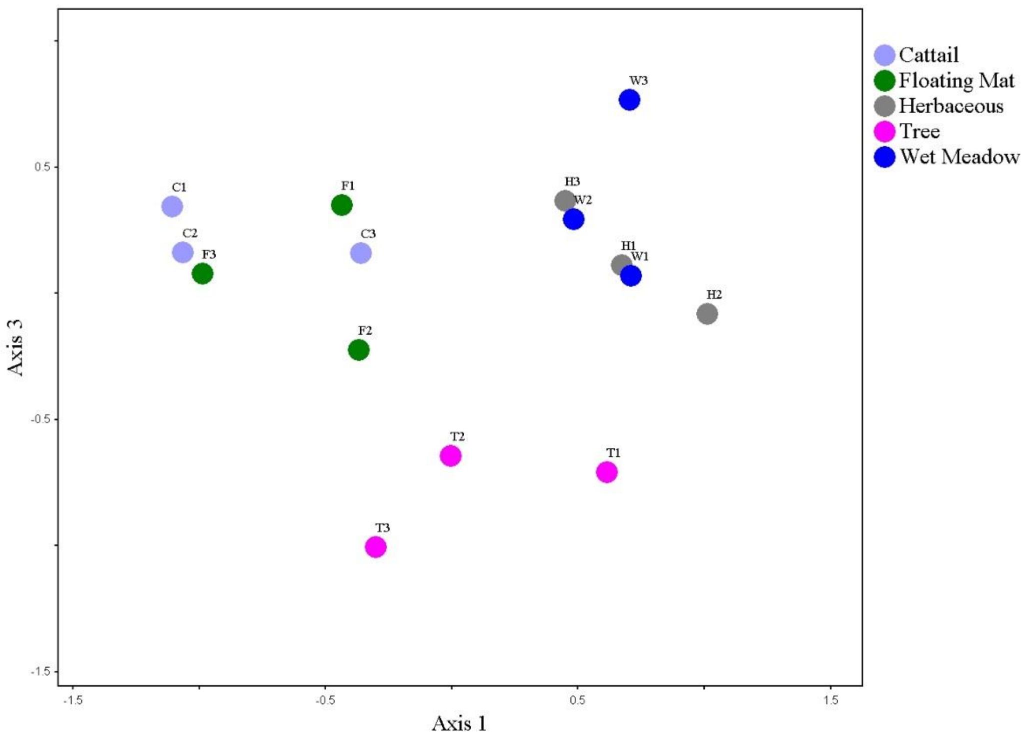

Figure 2.5. Nonmetric multidimensional scaling ordination of relative cover by vegetative community, Axis 1 versus Axis 3. Each symbol represents a single site.

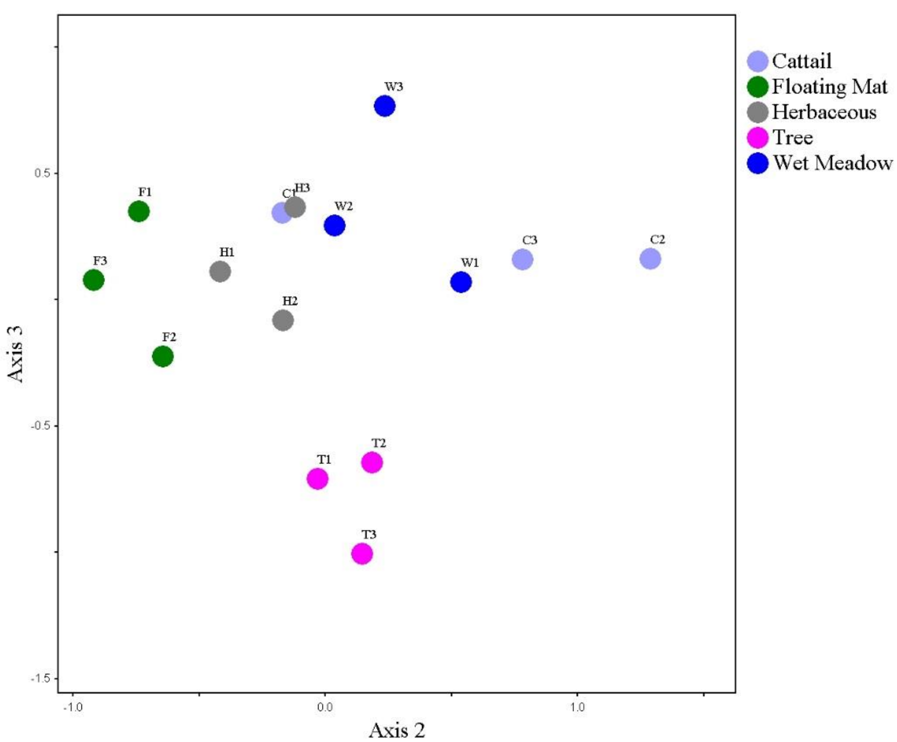

Figure 2.6. Nonmetric multidimensional scaling ordination of relative cover by vegetative community, Axis 2 versus Axis 3. Each symbol represents a single site.

Table 2.1. P-values from multiple-response permutation procedure (MRPP). Comparisons between the relative cover of plant species within each vegetative community (P-values adjusted for multiple comparisons using Benjamini-Hochberg correction).

Comparison			**P-value**
Cattail	vs.	Floating Mat	0.035
Cattail	vs.	Herbaceous	0.031
Cattail	vs.	Tree	0.031
Cattail	vs.	Wet Meadow	0.031
Floating Mat	vs.	Herbaceous	0.031
Floating Mat	vs.	Tree	0.031
Floating Mat	vs.	Wet Meadow	0.031
Herbaceous	vs.	Tree	0.031
Herbaceous	vs.	Wet Meadow	0.038
Tree	vs.	Wet Meadow	0.031

The average species richness (S) in the cattail communities was 26.3 (SE= 6.69) (Figure 2.7). The most common species in the cattail communities were *Typha x glauca* (55.3 %) and *Typha angustifolia* (13.4 %). The average S in the floating mat communities was 44.67 (SE= 6.33) (Figure 2.7) and *Carex emoryi* (42.5 %), *Carex aquatilis* (18.8 %), and *Eleocharis palustris* (10.6 %) were the most common species. The herbaceous communities had an average S of 55.67 (SE= 4.70) (Figure 2.7) and *Carex interior* (11.3 %) and *Helianthus nuttallii* (10.3 %) were the most common species. The tree communities had an average S of 53.00 (SE= 3.79) (Figure 2.7) and *Caltha palustris* (18.5 %) and *Salix bebbiana* (16.4 %) were the most common species. The wet meadow communities had an average S of 41.00 (SE= 4.04) (Figure 2.7) and *Glyceria striata* (29.5 %), *Deschampsia cespitosa* (13.5 %), and *Sonchus arvensis* (11.7 %) were the most common species. Average S was lower (ANOVA; $F_{4,10}$= 4.89; p= 0.02) in the cattail plant communities (S= 26.3) than in the herbaceous (S= 55.67) and tree communities (S= 53.00).

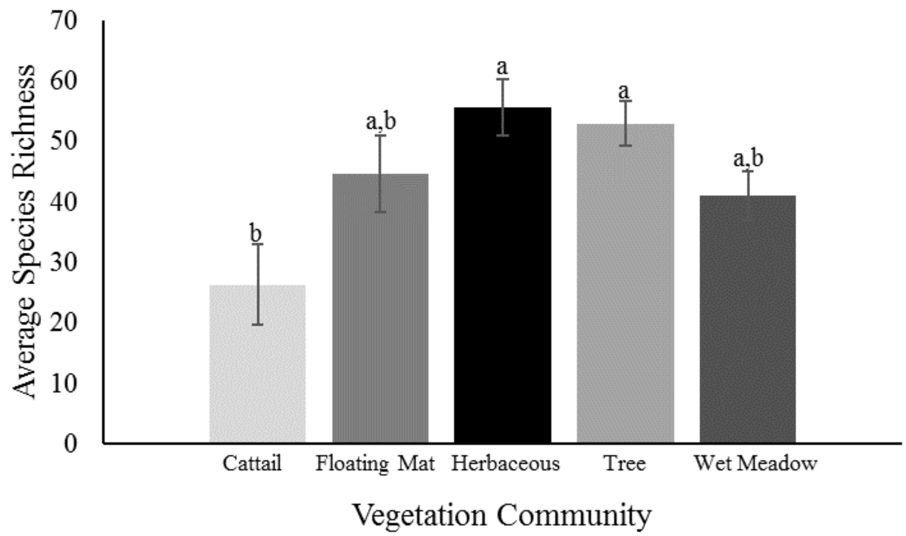

Figure 2.7. Average species richness by vegetative community. Lower case letters indicate significant differences (Tukey's HSD; $p < 0.05$).

The average evenness (E) was 0.36 (SE= 0.08) in the cattail communities, 0.48 (SE= 0.04) in the floating mat communities, 0.73 (SE= 0.02) in the herbaceous communities, 0.57 (SE= 0.05) in the tree communities, and 0.57 (SE= 0.06) in the wet meadow communities (Figure 2.8). Average E was lower (ANOVA; $F_{4,10}$= 5.85; p= 0.01) in the cattail communities (E= 0.36) than in the herbaceous communities (E= 0.73).

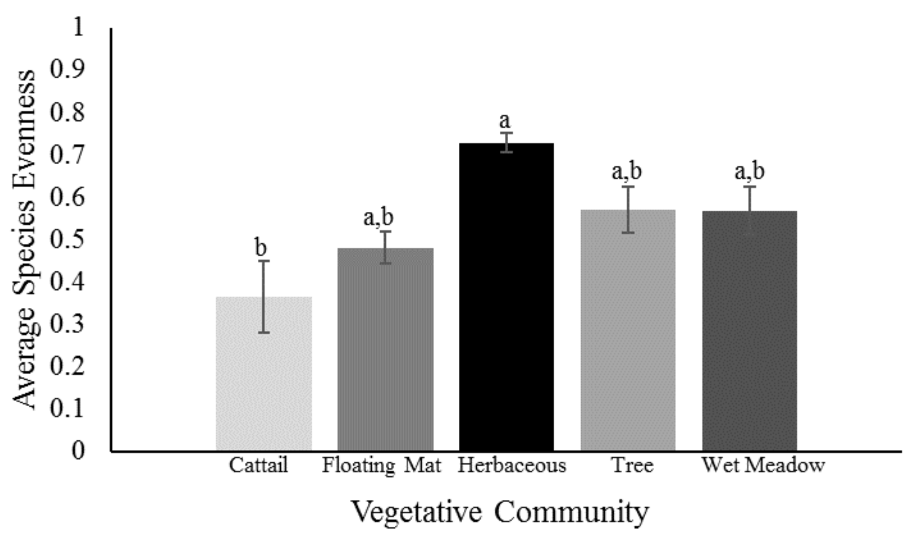

Figure 2.8. Average evenness by vegetative community. Lower case letters indicate significant differences (Tukey's HSD; $p < 0.05$).

The average Simpson Diversity Index (D) value for cattail communities was 0.47 (SE= 0.15), floating mats was 0.65 (SE= 0.04), herbaceous was 0.92 (SE= 0.01), tree was 0.80 (SE= 0.10), and wet meadow was 0.76 (SE= 0.04) (Figure 2.9). Average D was lower (ANOVA; $F_{4,10}$= 4.21; p= 0.03) in the cattail plant communities (D= 0.47) than in the herbaceous communities (D= 0.92).

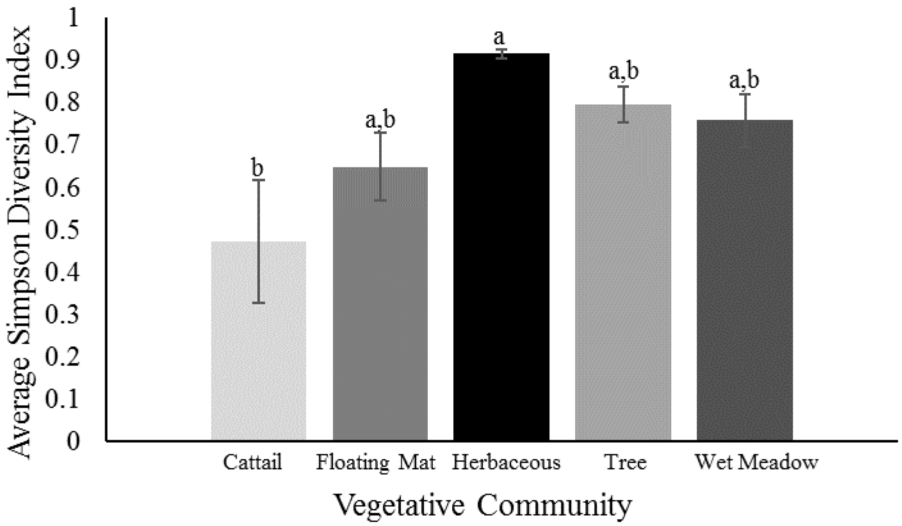

Figure 2.9. Average diversity by vegetative community. Lower case letters indicate significant differences (Tukey's HSD; $p < 0.05$).

The average relative cover of introduced species (% I) was higher in the cattail communities (68.96 %, SE= 12.49), than in the floating mat (0.39 %, SE= 0.06), herbaceous (12.33%, SE= 5.28), tree (12.33 %, SE= 10.40), and wet meadow (18.78 %, SE= 3.65) communities (Figure 2.10). Average % I was higher (ANOVA; $F_{4,10}$= 11.74; p= 0.0009) in cattail communities (% I= 68.96 %) than the other four communities.

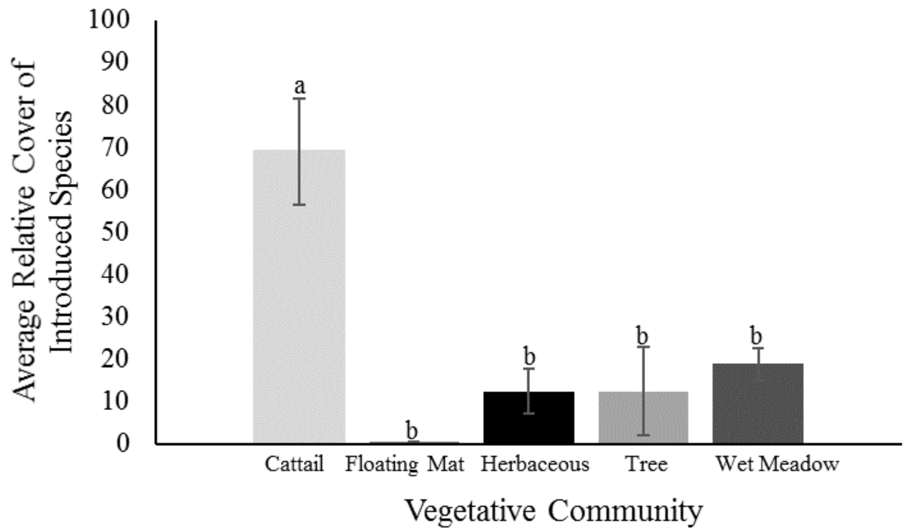

Figure 2.10. Average relative cover of introduced species by vegetative community. Lower case letters indicate significant differences (Tukey's HSD; $p < 0.05$).

Soil Description

We found two distinct horizons (O and A horizons) between 0 and 100 cm within the average cattail community soil profile (Figure 2.11A). The average lower boundary of the O horizon was 27 cm. The O horizon had a wet color of 2.5Y 2.5/1 and the A horizon had a wet color of 2.5Y 3/1. The texture of the O horizon was highly organic with granular structure, while the A horizon had a sandy clay texture with subangular blocky structure. We observed many very fine to fine roots and common very fine pores within 100 cm of the surface.

We found three distinct horizons (O_a1, O_a2, and A horizons) between 0 and 100 cm within the average floating mat community soil profile (Figure 2.11B). The average lower boundary of the O_a1 horizon was at 17 cm, with a second horizon break at 43 cm. The O_a1 horizon was colored 10YR 2/1 when wet. These horizons had granular structure with a highly organic texture. The O_a2 was colored 2.5Y 2.5/1 and the A horizon was 2.5Y 3/1 when wet. The O_a2 horizon had a mucky loam texture and granular structure and the A horizons had sandy loam textures and granular structures. We observed many very fine to fine roots and many fine pores within 100 cm of the surface.

We found three distinct horizons (A1, A2, and B horizons) between 0 and 100 cm within the average herbaceous community soil profile (Figure 2.11C). The average lower boundary for the A1 horizon was 33 cm and the lower boundary of the second horizon was on average at 79 cm. The A1 horizon was 2.5Y 2.5/1 colored when wet, the A2 horizon was 2.5Y 3/2, and the B horizon was 2.5Y 2.5/1 when wet. The A1 horizon ranged from sandy clay loam to silty clay loam in texture. The A2 horizon ranged from sandy loam in texture to silty clay loam. The first and A2 horizon had granular structure. The B horizon was sandy loam to silt loam and had subangular blocky structure. We found many very fine roots and pores in the A1 horizon, and few very fine roots and many very fine pores in the A2 and B horizons.

We found two horizons (O and A horizons) from 0 to 100 cm within the average tree community soil profile (Figure 2.11D). The lower boundary of the surface horizon was 25 cm. The O horizon was 10YR 2/1 colored and the A horizon was 2.5Y 2.5/1 when wet. The O horizon was a highly organic texture with granular structure. The A horizon was a sandy loam texture with subangular blocky structure. We found many very fine roots and pores in the O horizon and common very fine roots and pores in the A horizon.

We found three horizons (A1, A2, and B horizons) from 0 to 100 cm within the average wet meadow community soil profile (Figure 2.11E). The average lower boundary was at 42 cm for the A1 horizon and the A2 horizon had lower boundaries around 68 cm. The wet color for the A1 horizon was 10YR2/1, the A2 horizon 2.5Y2.5/1, and the B horizon 2.5Y 2.5/1. The A1 horizon had granular structure and ranged from a silt loam to a clay loam in texture. The A2 horizon was subangular blocky in structure and ranged from a silt loam to a silty clay in texture.

The B horizon had a prismatic structure and had a silty clay texture. We found many very fine roots and pores in the A1 horizon. The A2 horizon had common very fine roots and many very fine pores. The B horizon had few very fine roots and common very fine pores.

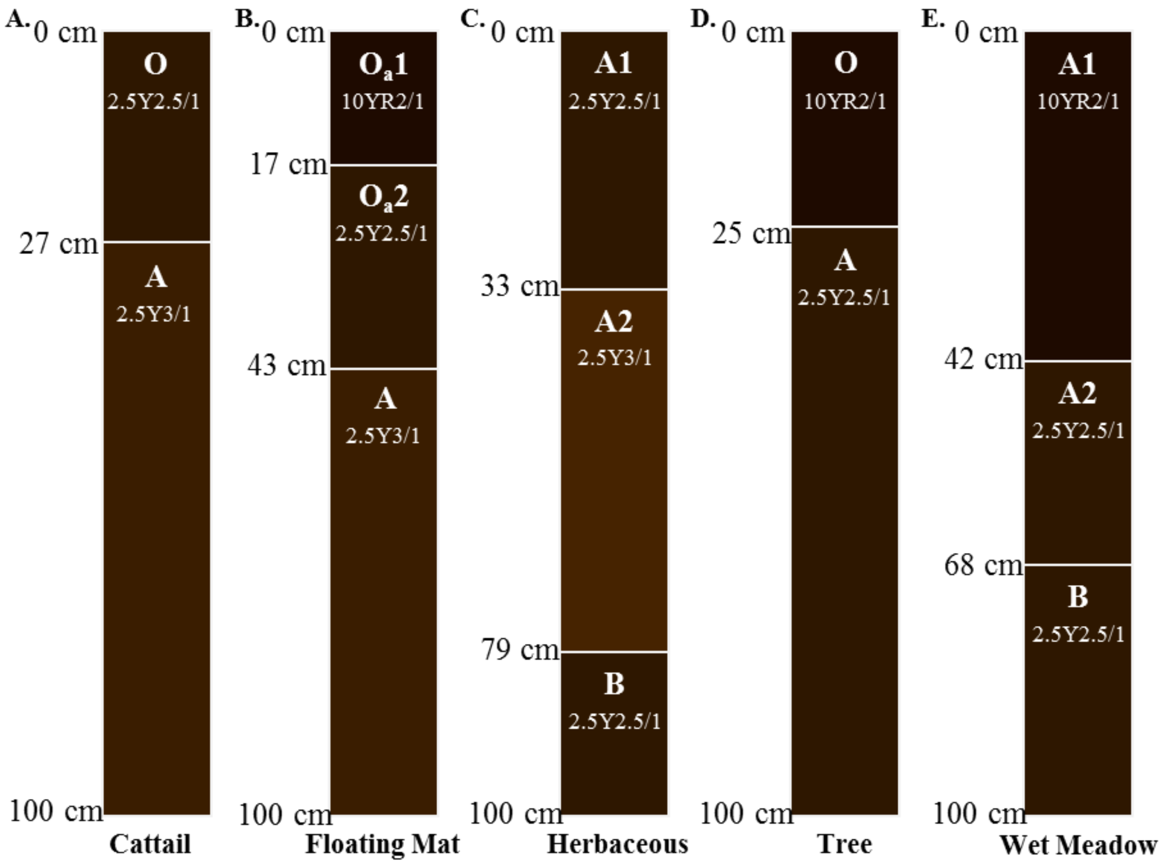

Figure 2.11. Average soil profile by vegetative community including horizon breaks and wet soil color.

Soil Chemistry

Percent organic matter (% OM) was not significantly different among our five vegetative communities (ANOVA; $F_{4,10}$= 3.42; p = 0.05). The average % OM in the surface horizon was 31.4% (SE= 5.89) in the cattail communities, 23.73% (SE= 1.32) in the floating mat communities, 13.30% (SE= 2.71) in the herbaceous communities, 20.70% (SE= 5.02) in the tree communities, and the 16.10% (SE= 1.93) in the wet meadow communities (Figure 2.12).

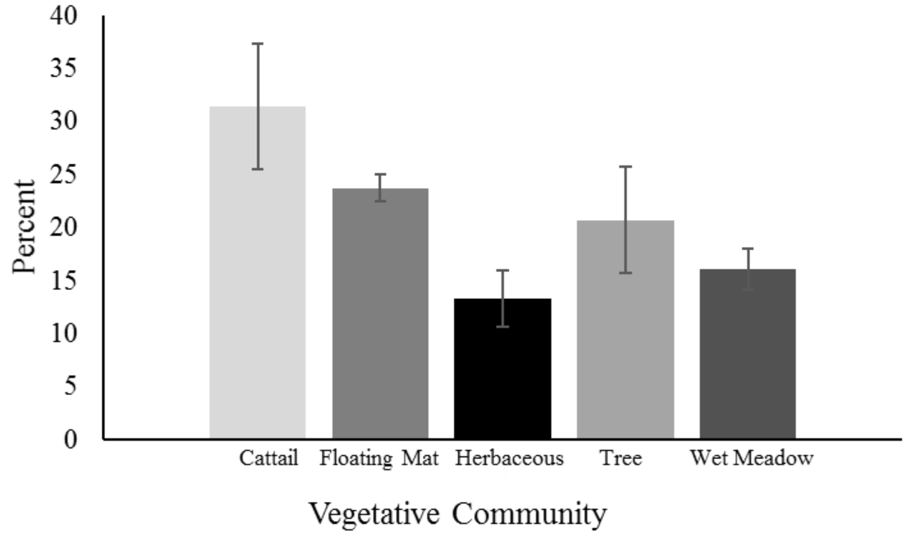

Figure 2.12. Average % OM in the surface horizon by vegetative community. There were no significant differences in % OM among our five wetland plant community types.

pH was not significantly different among our five vegetative communities (ANOVA; $F_{4,10}$= 1.61; p= 0.25). The average pH in the surface horizon was 7.23 (SE= 0.12) in the cattail communities, 7.67 (SE= 0.18) in the floating mat communities, 7.57 (SE= 0.19) in the herbaceous communities, 7.37 (SE= 0.03) in the tree communities, and 7.67 (SE= 0.19) in the wet meadow communities (Figure 2.13).

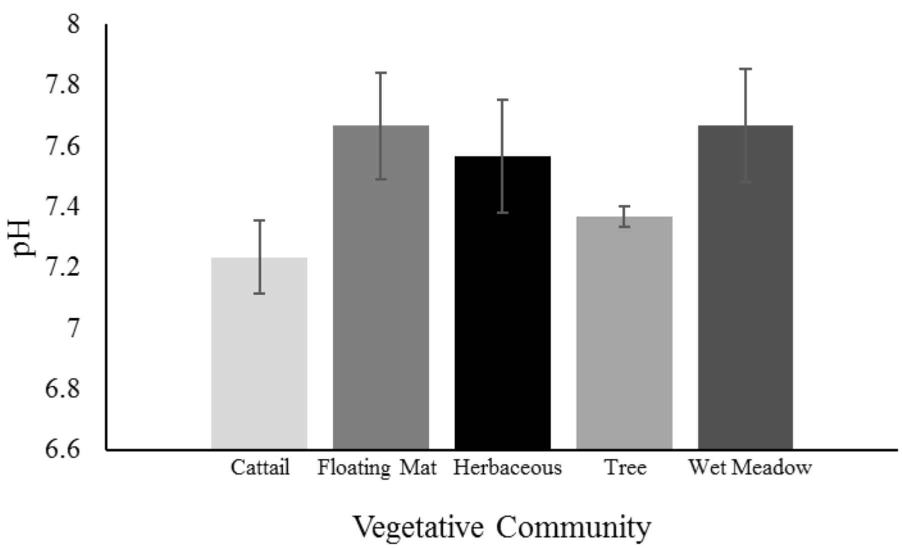

Figure 2.13. Average pH value in the surface horizons by vegetative community. There were no significant differences in pH among our five wetland plant community types.

Electrical conductivity (EC) was not significantly different among our five vegetative communities (ANOVA; $F_{4,10}$= 2.68; p= 0.09). The average EC in the surface horizon was 1.47 dS/m (SE= 0.13) in the cattail communities, 1.08 dS/m (SE= 0.11) in the floating mat communities, 1.06 dS/m (SE= 0.17) in the herbaceous communities, 1.27 dS/m (SE= 0.09) in the tree communities, and 0.99 dS/m (SE= 0.06) in the wet meadow communities (Figure 2.14).

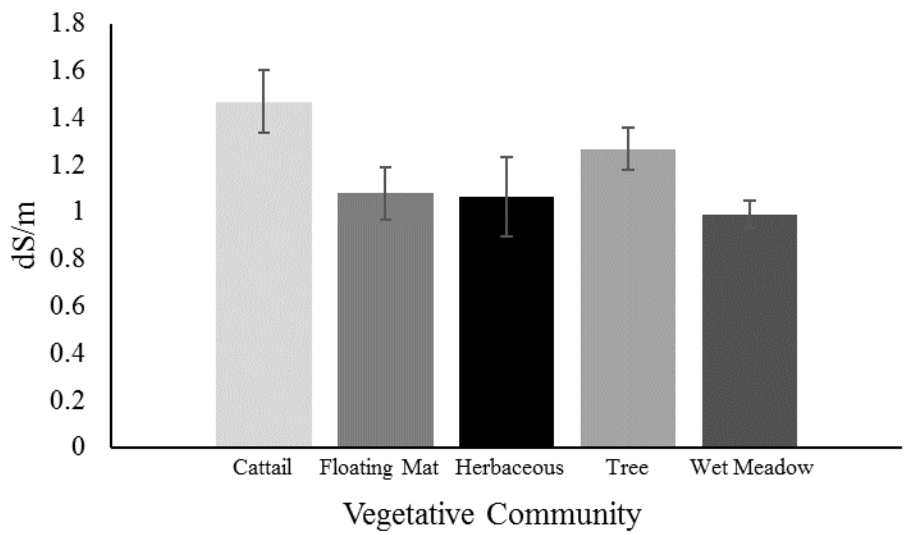

Figure 2.14. Average EC values in the surface horizon by vegetative community. There were no significant differences in EC among our five wetland plant community types.

There were no hummocks present within the cattail vegetative communities. The average hummock height was 16.96 cm within the floating mat communities, 17.52 cm within the herbaceous communities, 7.78 cm within the tree communities, and 20.65 within the wet meadow communities (Table 2.2). At the time of sampling, the average depth of water was +1.5 cm within the cattail communities, -8.7 cm within the floating mat communities, -20.7 cm within the herbaceous communities, -19.3 cm within the tree communities, and -15.0 cm within the wet meadow communities (Table 2.3).

Table 2.2. Average hummock height by vegetative community.

Vegetative Community	Average height (cm)	Max (cm)	Min (cm)	n
Cattail	0	0	0	0
Floating Mat	16.96	28	8	20
Herbaceous	17.52	32	8	26
Tree	7.78	16	0	8
Wet Meadow	20.65	37	11	25

Table 2.3. Average depth to water by vegetative community. + is above the surface, - is below the surface.

Vegetative Community	Average depth (cm)
Cattail	+1.5
Floating Mat	-8.7
Herbaceous	-20.7
Tree	-19.3
Wet Meadow	-15

Discussion

Vegetation

The NMS and MRPP analysis of all 150 species in the sampled plots determined that all five vegetative communities were distinct. Across the fen, the average species richness was 44.1 species per 10 square meter plot. The cattail communities on average had lower S value than the other communities, although there was no significant difference in S between the cattail communities and the floating mat and wet meadow communities. The cattail communities also had lower E and D values than the other communities, although these variables were only significantly different from the herbaceous communities. The significant difference between the cattail communities and other vegetative communities determined by the MRPP analysis and the ANOVA analysis of the vegetation data could be a result of hydrology. The cattail communities were generally found in the drainage areas of the site, where water ponded for the longest period during the growing season. The cattail communities also had the deepest water when we sampled water depth. Areas that are more frequently flooded often have lower species diversity compared to areas with intermediate flooding (Pollock et al. 1998).

We determined the NMS Axis 1 was correlated with the individual species most commonly found in the cattail and floating mat communities. Both of these communities had deeper water at the time of sampling, which could indicate increased water ponding resulting conditions that allowed particular species to persist in these areas. Saturated conditions often limit nutrient availability, which can result in the presence of specialized vegetation (Almendinger & Leete 1998b; Turner et al. 2000; Amon et al. 2002; Bedford & Godwin 2003)

The lower species richness, evenness, and diversity values found in the cattail communities may also be a result of the significantly higher percentage of introduced species compared to the other communities. Introduced vegetation can change the structure of the vegetative community (Levine et al. 2003; Vila et al. 2011; Larken et al. 2012) by reducing the fitness and growth of native species and decrease species richness and diversity (Vila et al. 2011). Introduced species, such as *T. glauca*, often form dense monotypic or nearly monotypic stands, as they rapidly spread via clonal reproduction leaving behind large amounts of litter. The increased litter produced by *T. glauca* results in cooler soil temperatures which can delay soil thaw and decrease light availability limiting seed germination and primary productivity of the native vegetation (Larkin et al. 2012). The increased litter within the cattail communities at our site could limit the habitat potential for other native vegetation resulting in the reduced species richness, evenness, and diversity.

Introduced species can outcompete native vegetation and alter carbon and nitrogen cycles. In a study of the Cheboygan Marsh in Michigan, researchers found decreased plant diversity in areas with *Typha x glauca* (hybrid cattail), compared to the areas dominated by native vegetation. This study also found increased nutrients in areas dominated by *T. glauca* (Angeloni et al. 2006). High species diversity in fens is thought to be associated with the nutrient poor conditions (Almendinger & Leete 1998b; Turner et al. 2000; Bedford & Godwin 2003), therefore, increased nutrients due to the presence of *T. glauca* could account for decreased species richness and diversity at these sites.

Floating mat communities had moderate S, E, and D values, and a very low % I. These areas, like the cattails communities, have water for a majority of the growing season. Instead of being drainage ways, the floating mat communities have water near the surface and are situated in areas where groundwater pools. The relatively constant presence of groundwater could account for the low S, E, and D values, as the increased frequency of flooding is often correlated with decreased diversity (Pollock et al. 1998). Deeper water creates stressful environments for vegetation (Pollock et al. 1998), and can result in lower species diversity compared to the herbaceous and wet meadow communities where the entire area is less often completely flooded.

Although the MRPP analysis found the floating mat communities to be significantly different from all other communities sampled, the vegetation variables tested through ANOVA did not yield significant differences from the other communities. Therefore, factors not tested may account for the difference. Climate and geology effect ecological condition including the vegetative and soils properties (Nekola 2004) and thus could be factors that account for the variation. Groundwater moving through geologic deposits accumulates minerals and nutrients, which can alter soil and vegetative properties (Vitt & Chee 1990; Almendinger & Leete 1998a; Turner et al. 2000; Bedford & Godwin 2003). One of these factors may be correlated with the

NMS Axis 2 as we determined this Axis to be negatively correlated with the species most commonly found in the floating mat communities.

The herbaceous communities had the highest S, E, and D values compared to the other communities, although there were only significant differences between the herbaceous and cattail communities for these three measures. The high diversity relative to the other vegetative communities may be a result of the frequency these areas experience flooding. While the other areas were found in zones of water drainage (i.e. cattails) and near the seepage point (i.e. trees, floating mats), the herbaceous communities were found further away from direct areas of water ponding and flow through areas. However, the herbaceous communities were found in areas that will experience ponding during part of the growing season. High species diversity is often correlated to the frequency of flooding and the spatial variation in flood frequency, where areas with intermediate flood frequencies will have higher species diversity (Pollock et al. 1998).

The significant difference between the herbaceous communities and all other communities found with the MRPP analysis may be a result of microtopography and flood frequency. Microtopography can affect the spatial variation in flood frequency, resulting in increased species diversity with increased variations in microtopography (Pollock et al. 1998). The herbaceous and wet meadow communities had the greatest variation in microtopography with hummock height ranging from 8 cm to 32 cm and 11 cm to 37 cm respectively. In addition, graminoid vegetation has been found to respond to spatial variation in flood frequency compared to other vegetation types (Pollock et al. 1998), which could account for higher diversity within the herbaceous communities compared to the other vegetative communities.

The tree communities had high S values, with no significant difference from the floating mat, herbaceous, or wet meadow communities. The tree communities also had moderate E and D values with low % I. The high S value could be attributed to the adaptation of fen specialist species to waterlogged and nutrient poor conditions (Almendinger & Leete 1998b; Turner et al. 2000; Bedford & Godwin 2003). The low % I could also be a result of the waterlogged conditions, where saturated conditions in areas of groundwater discharge reduce nutrient availability and can limit the growth potential of vegetation (Turner et al. 2000; Visser et al. 2000; Glenz et al. 2006). The tree communities may be too wet and too nutrient poor to support the introduced species found in other vegetative communities across the fen. The moderate E and D values may also be a result of the fitness of individual species in waterlogged conditions.

We determined the NMS Axis 3 to be correlated with the species most commonly found in the tree communities. The significant difference between the tree communities and the other vegetative communities from the MRPP analysis may be a result of the waterlogged conditions. These communities were dominated by *Caltha palustris* and *Salix bebbiana* which are successful in saturated conditions because of their biological adaptations. The roots of *C. palustris* have been found to increase in diameter in waterlogged situations due to the presence of aerenchyma cells (Visser et al. 2000). *Salix* spp. are well adapted for waterlogged conditions with their coarse bark, which allow the trees to maintain useable oxygen (Glenz et al. 2006). These adaptations allow these species, as well as other wetland and fen species to survive in waterlogged situations with poor oxygen availability (Visser et al. 2000).

The significant difference between the tree communities and the other vegetative communities may also be a result of landscape position. Fens often lack tree or shrub communities and usually are dominated by graminoid vegetation. However, when tree or shrub communities are present they are generally stunted and isolated along the margins of the fen (Malmer 1986l; Aaseng et al. 2005). At our site, the tree communities were found along the hillslope surrounding the major portion of the fen. These communities maybe significantly different from the other communities because of their position on the landscape and how the soil developed due to water movement on and through the slopes.

The wet meadow communities had moderate S, E, and D, and % I values. These variables tested did not account for the significant difference between the wet meadow communities and the other vegetative communities determined by our MRPP analysis. The differences determined in the MRPP analysis could be a result of flood frequency and microtopography. These communities are in areas that are less frequently flooded than the cattail, floating mat, and tree communities, but more frequently flooded than the herbaceous communities. The moderate measurements for diversity could be a result of flood frequency. The vegetation in the wet meadow environments may be more stressed than the herbaceous communities and less stressed than the cattail, floating mat, and tree communities. The flood frequency along with the microtopography may create local environments where more introduced species do not need adaptations to survive in waterlogged conditions. The introduced species have the opportunity to become established on the drier top of the hummocks. The hummocks create microtopography to reduce the flood frequency of the entire wet meadow area which can lead to higher species diversity (Pollock et al. 1998), but may also act as a buffer for the introduced species.

The North Dakota Natural Heritage Program proposed a list of species to be considered priorities for conservation in North Dakota. Species are ranked as Level I, II, or III to prioritize conservation efforts. Six species found at our study site are on this proposed list. *Cypripedium candidum* is a Level I species, *Cypripedium parviflorum* and *Carex sterilis* are Level II species, and *Parnassia palustris*, *Rhynchospora capillacea* and *Utricularia intermedia* are Level III species (North Dakota Natural Heritage Program 2013). The presence of these species within our study area could make this site a priority for conservation in North Dakota.

Soil

All five vegetative communities had dark colors in the surface horizon. Dark soil colors often indicate a high percent of organic matter (Davis & Lucas 1959). The saturated conditions result in anoxic conditions and slow decomposition allowing organic matter to accumulate (Amon et al. 2002; Schaetzl & Anderson 2005; Mitsch & Gooselink 2015). The more saturated vegetative communities (cattail, floating mat, and tree communities) had greater % OM at the surface and had highly organic textures. The accumulation of organic matter due to reduced rates of decomposition results in increased porosity, increasing the potential for water movement through the soil (Boelter 1969; Carey et al. 2007). Organic soils have 20 to 35 percent organic matter (Davis & Lucas 1959; Mitsch & Gooselink 2015), thus cattail, floating mat, and tree communities met this criterion at our study site.

Soil structure influences water movement, nutrient cycles, carbon sequestration, and root penetration (Bronick & Lal 2005). The surface horizons in all five vegetative communities were granular structure. Soils with granular structure are porous and have high permeability (Schaetzl & Anderson 2005), improving the ability of water to move through the soil profile (Carey et al. 2007). Soil structure affects the movement and retention of water within the soil as well as root penetration. The soils sampled in our five vegetative communities had relatively loose structure, which promotes water movement and plant root growth (Bronick & Lal 2005).

pH values ranged from 7.23 to 7.67 across the fen site. Fens often have higher species diversity because of the moderate pH values (Jassey et al. 2014). The variations in pH values could be a result of variations in water levels among the five vegetative communities. Malmer (1986) found soil properties like pH as well as the variations in vegetative communities across a single site varied based on changes in water levels. Plants are generally tolerant of particular pH ranges, generally varying from weakly acidic to weakly alkaline, where some species are tolerant of more acidic or more basic environments (Larcher 1995). Vegetation is sensitive to pH because biochemical processes within plants often function optimally with pH values between six and seven. Soil pH also affects the availability of nutrients to the vegetative communities. Acidic soils will have free aluminum, iron, and manganese ions and be depleted of calcium, magnesium, potassium, and phosphate ions, limiting nutrient availability in these environments. Calcareous soils, like the soils found at our study site, will be limited by the availability of iron, phosphorus, and manganese (Larcher 1995).

EC is an indicator for soil health and measures the amount of salts in the soil. High concentrations of salts can limit the nutrient availability and soil microorganism activity, as well as disrupt the water balance in the soil (USDA 2014). The EC values at our site ranged from 0.99 to 1.47 dS/m. EC values less than 1.0 dS/m are considered non-saline and microbial processes as well and plant development will not be hindered. EC values above 1.0 dS/m can inhibit microbial processes and result in reduced soil health, however each species has a saline threshold so some species will be able to persist in areas with higher EC values (USDA 2014). The soils at our study site range from non-saline to slightly saline. Likely, the vegetative communities present at these sites are adapted to the available nutrients and saline conditions, allowing them to persist in the particular environments across the site.

Although the MRPP analysis determined all five vegetative communities to be significantly different, none of the soil properties tested were found to differentiate the vegetative communities. Therefore, the variations in vegetation across the fen may be accounted for by spatial variations in microtopography and flood frequency. Soil structure and other soil chemistry variables, hydrology, microclimate, and underlying geologic deposits may also cause distinct vegetative communities across the landscape. However, variations across a single site make it challenging to determine the specific characteristics that distinguish one community from another (Pollock et al. 1998).

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APPENDIX A. NORTH DAKOTA RAPID ASSESSMENT METHOD FOR WETLANDS FROM HARGISS 2009

Directions:

The NDRAM for wetlands was created to rapidly assess temporary, seasonal, and semi-permanent wetlands in the Prairie Pothole Region based on the plant communities present. Results of the NRDAM should indicate results similar to the Index of Plant Community Integrity (IPCI) (DeKeyser 2000, DeKeyser et al. 2003, Hargiss 2005, and Hargiss et al. 2008).

Before conducting the NDRAM employees should complete the short NDRAM field training course. This course will teach them the methods involved in the NDRAM, how to identify significant characteristics of the wetland, and the basic plant community information needed to properly use the NDRAM. Additional training on the HGM Model and the IPCI may also be helpful, but not necessary, to complete the NDRAM. Another additional resource that may be helpful is Stewart and Kantrud (1971).

The NDRAM can be completed by anyone who has had the short field course. The NDRAM should be used as an indicator of wetland condition in an area. However, further investigation into plant communities present and land use practices will be helpful in making recommendations for management of an area. The NDRAM can be used every few years to indicate change in wetland condition. When combined with the IPCI over a larger area, regional wetland plant community trends can also be determined.

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North Dakota Rapid Assessment Method Form:

Site Name__________________________ **Date**___________________________

Land Ownership____________________

Person(s) assessing wetland____

Legal Description____

County____

GPS Information:

Datum_____________

N__________________

W__________________

General Site Description____

____

____

____

____

____

____

____

____

Photo's

Photo Number	Direction Facing	Description

Use space below to draw a detailed picture of the wetland. Be sure to include different groups of vegetation and any distinct features. Create a legend for your map. Circle the % cover of the different types of plants on the right.

Sedges	**0-25%**	**25-50%**	**50-75%**	**75-100%**
Cattails	**0-25%**	**25-50%**	**50-75%**	**75-100%**
Grasses	**0-25%**	**25-50%**	**50-75%**	**75-100%**
Rushes	**0-25%**	**25-50%**	**50-75%**	**75-100%**
Forbs	**0-25%**	**25-50%**	**50-75%**	**75-100%**
Shrubs	**0-25%**	**25-50%**	**50-75%**	**75-100%**
Trees	**0-25%**	**25-50%**	**50-75%**	**75-100%**
Other:_______	**0-25%**	**25-50%**	**50-75%**	**75-100%**

N ↑

1 square = ____ m

Overall wetland is approximately ________m X ________m

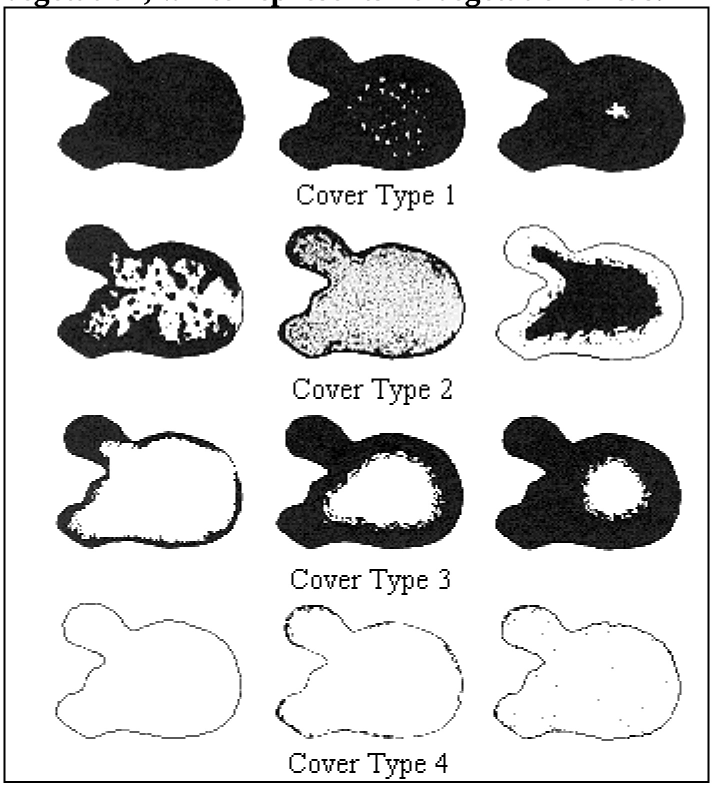
Hydrologic classification (temporary, seasonal, etc.) ___________________________

Site Characterization:

Estimate amount of standing water:

Total wetland area covered by standing water	0	1-25	26-50	51-75	76-100
If water is present:					
Percentage of water <1 ft. deep	0	1-25	26-50	51-75	76-100
Percentage of water 1-3 ft. deep	0	1-25	26-50	51-75	76-100
Percentage of water >3 ft. deep	0	1-25	26-50	51-75	76-100

Estimate (by circling picture below) amount and distribution of cover. Black represents vegetation, white represents no vegetation areas.



Land use and disturbances (check all that apply):

	Dugout		Haying
	Road/prairie trail		Drought
	Cropping		Restored/Reclaimed
	Drain		Idle
	Grazed		Other_________________________

Wetland Classification:

Poor Condition: Poor condition wetlands are wetlands that are highly disturbed with low functioning (Example: cropped, drained, etc.).

Fair Condition: Fair condition wetlands are wetlands that have been disturbed in the past or are currently moderately disturbed. They perform many wetland functions, but are not at full potential compared to less disturbed native wetlands (Example: hayed, mowed, CRP, etc.).

Good Condition: Good condition wetlands are native properly functioning wetlands that are for the most part undisturbed (Example: grazed, native areas).

Preliminary Observations:

#	Question	Circle One	
1	Critical Habitat. Is the wetland in an area that has been designated by the U.S. Fish and Wildlife Service as "critcal habitat" for any threatened and endangered species?	Yes Wetland should be evaluated for possible Good condition status.	No
2	Critical Habitat. Is this wetland a fen or does it contain a fen?	Yes Wetland should be evaluated for possible Good condition status.	No
3	Threatened or Endangered Species. Is the wetland known to contain an individual of, or documented occurrences of, federal or state-listed threatened or endangered plant or animal species?	Yes Wetland should be evaluated for possible Good condition status.	No
4	Poor Condition Wetland. Is the wetland completely plowed through all zones on a regular basis and planted with a crop?	Yes Wetland is a poor condition wetland.	No
5	Good Condition Wetland. Is the wetland in an area that has never been disturbed other than light-moderate grazing, and contains mostly native perennial species?	Yes Wetland should be evaluated for possible Good condition status.	No

Metrics

Metric 1. Buffers and surrounding land use.

1a. Calculate Average Buffer Width

Score	Rating Description
	WIDE. Buffer averages 50m or more around wetland perimeter (10pts)
	MEDIUM. Buffer average 25m to <50m around wetland perimeter (7 pts)
	NARROW. Buffer averages 10m to <25m around wetland perimeter (4 pt)
	VERY NARROW. Buffer averages <10m around wetland perimeter (0 pts)
	OTHER.

1b. Intensity of Surrounding Land Use. Select one or more, average the scores.

Score	Rating Description
	VERY LOW. Native prairie, light to moderate grazing, etc. (10 pts)
	LOW. Hayed prairie area, CRP, etc. (7 pts)
	MODERATELY HIGH. Farm, conservation tillage, planted alfalfa (4 pts)
	HIGH. Urban, row cropping, etc (1 pt)
	OTHER.

	Total for Metric 1 (out of possible 20).

Metric 2. Hydrology, Habitat alteration, and Development.

2a. Substrate/Soil Disturbance. This metric evaluates physical disturbances to the soil and surface substrates of the wetland. The labels on the categories are intended to be descriptive but not controlling. Examples of disturbance include: filling, grading, plowing, hoove action, vehicle use, sedimentation, dredging, etc.

Score	Rating Description
	NONE. There are no disturbances, or beneficial disturbances Ex. light to moderate grazing and fire (7 pts).
	RECOVERED. The wetland appears to have recovered from past disturbances (5 pts).
	RECOVERING. The wetland appears to be in the process of recovering from past disturbances (3 pts).
	RECENT OR NO RECOVERY. Complete removal of vegetation and soil exposed, the disturbances have occurred recently, and/or the wetland has not recovered from past disturbances, and/or the disturbances are ongoing (1 pt).
	OTHER

2b. Plant Community and Habitat Development. This metric asks the rater to assign an overall rating of how well-developed the wetland is in comparison with other ecologically or hydrogeomorphically similar wetlands; based on the quality typical of the region.

Score	Rating Description
	EXCELLENT. Wetland appears to represent best of its type or class. Ex. the wetland is found on native prairie and appears to be diverse in native plant species. (12 pts)
	VERY GOOD. Wetland appears to be a very good example of its type or class but is lacking characteristics which would make it excellent. Ex. wetland may be on native prairie but is lacking diversity because of being left idle or herbicide application. (10 pts)
	GOOD. Wetland appears to be a good example of its type or class but because of past or present disturbances, successional state, or other reasons, it is not excellent. (8 pts)
	MODERATELY GOOD. Wetland appears to be a fair to good example of its type or class. Ex. wetland has past disturbances such as heavy grazing, restoration, or draining that have affected the area. (6 pts)
	FAIR. Wetland appears to be a moderately good example of its type or class, but because of past or present disturbances, successional state, etc. it is not good. Ex. a combination of native and non-native portions to the wetland with low diversity of plant species. (4 pts)
	POOR TO FAIR. Wetland appears to be a good to fair example of its type or class. Ex. wetland may be a monoculture of one plant species or may have native species in a buffer around the wetland, but outer zones are cropped. (2 pts)
	POOR. Wetland appears to not be a good example of its type or class because of past or present disturbances, successional state, etc. Ex. wetland may be completely cropped through with no perennial plant community present. (0 pt)

2c. Habitat Alteration and Recovery from Current and Past Disturbances. This metric evaluates the disturbance level of wetland habitat and the ability to recover from habitat alterations. Ideal management involves some form of disturbance such as moderate grazing or fire to maintain plant vigor and diversity. Leaving areas idle and haying can lead to a monoculture of species. Restored and CRP areas take time to become properly functioning communities and are often planted with at least partially non-native species.

Score	Rating Description
	MOST SUITABLE. The wetland appears to have recovered from past alterations and alterations have been beneficial to habitat. (10 pts).
	NONE OR NONE APPARENT. There are no alterations, or no alterations that are apparent to the rater (7 pts).
	RECOVERING. The wetland appears to be in the process of recovering from past alterations (4 pts).
	RECENT OR NO RECOVERY. The alterations have occurred recently, and/or the wetland has not recovered from past alterations, and/or the alterations are ongoing (1 pt).
	OTHER.

2d. Management.

	Fire or Moderate Grazing. If the area has been burned or is moderately grazed at proper intervals. (4 pts)
	Restored, CRP, Hayed, or Idle. If the area is restored, hayed, planted with CRP, left idle, or has large buffer before cropping begins. (2 pts)
	Cropped. If the wetland is cropped through or cropped with only a very narrow buffer. (0 pts)
	OTHER.

2e. Modifications to Natural Hydrologic Regime. This question asks the rater to identify alterations to the hydrologic regime of the wetland (ex. ditches, drains, etc.) and the amount of recovery from such alterations.

Score	Rating Description
	NONE. There are no modifications or non modifications that are apparent to the rater (12 pts).
	RECOVERED. The wetland appears to have recovered from past modifications to the fullest extent possible. Ex. long established road (8 pts).
	RECOVERING. The wetland appears to be in the process of recovering from past modifications (4 pts).
	RECENT OR NO RECOVERY. The modifications have occurred recently, and/or has not recovered from past modifications, and/or the modifications are ongoing (1 pt).
	OTHER.

2f. Potential of Wetland to Reach Reference (Native) Condition for the Area. This question asks the rater to use their best professional judgment and determine the condition of the wetland and whether it is trending in a positive or negative direction (questions 2a – 2e may help in this determination). In this metric reclamation refers to taking off soil and replacing with wetlands soils and seed bank (strip mining), restoration involves seeding and management of wetland area, management includes a management system such as light to moderate grazing and/or fire and may include spraying of unwanted species.

Score	Rating Description
	EXCELLENT. Wetland is at or near reference condition (12 pts).
	GOOD POTENTIAL. Wetland is disturbed in some way so not at reference condition, but could achieve reference condition easily over time (10 pts).
	MODERATE POTENTIAL. Wetlands is disturbed, but with proper management and time it could return to reference condition (7 pts).
	MODERATELY POOR POTENTIAL. Through proper management and potential restoration/reclamation the wetland may return to reference condition. (5 pts).
	POOR POTENTIAL. Minor potential for return to reference condition, but restoration/reclamation would be needed (2 pt).
	NO POTENTIAL. No potential for return to reference condition without extreme restoration/reclamation efforts (0 pts).

	Total for Metric 2 (out of possible 57).

Metric 3. Vegetation

3a. Invasive species (include in estimate of 3m buffer of low prairie zone). Amount of aerial plant covered by invasive species. Invasive species (native or non-native) include but are not limited to brome, reed canary, quack, Kentucky bluegrass, and crested wheat grasses, as well as Canada thistle and leafy spurge. Annual crops and weeds should be considered invasive.

Score	Rating Description
	ABSENT. (3 pts)
	NEARLY ABSENT. <5% aerial cover of invasive species (1 pt)
	SPARSE. 5-25% aerial cover of invasive species (0 pt)
	MODERATE. 25-75% aerial cover of invasive species (-1 pts)
	EXTENSIVE. >75% aerial cover of invasive species (-3 pts)

3b. Overall condition of wetland based on plant species using best professional judgment from professional wetland botanist. Walk around wetland area making mental note of plant species present, variety, abundance, etc.

Score	Rating Description
	VERY GOOD (20 pts). Undisturbed native area with a variety of plant species throughout wetland (grasses, sedges, rushes, forbs, etc). Moderate grazing may be utilized. No major impairments to area.
	GOOD (15 pts). Area is still relatively native with a good variety of species. There is an impairment (road, haying, spraying, etc) that has affected the condition of the wetland.
	FAIR (10 pts). Area has been impaired either in the past and is recovering or is currently being impaired but not by something that would decimate the plant community. (CRP, haying, etc.)
	POOR (5 pts). Area is heavily disturbed but there are some plant species still intact. Plant community will consist mostly of non-native annual species, but there may be some native or perennials present. Large populations of invasive species may be present.
	VERY POOR (0 pt). Wetland is heavily disturbed (cropping, hayland, etc) and the plant community if one exists consists of mostly non-native annual species with very little variety. Invasive species may dominate the plant community.

	Total for Metric 3 (out of possible 23).

TOTAL.

Score	
	Total from Metric 1.
	Total from Metric 2.
	Total from Metric 3.
	Rapid Assessment Score

APPENDIX B. SPECIES LIST BY VEGETATIVE COMMUNITY IN EDDY COUNTY FEN

Community	Species	Physiognomy	Native/Introduced
Cattail			
	Asclepias incarnata	forb	native
	Calamagrostis stricta	grass	native
	Caltha palustris	forb	native
	Carex aquatilis	sedge	native
	Carex hystericina	sedge	native
	Carex interior	sedge	native
	Carex lanuginosa	sedge	native
	Carex prairea	sedge	native
	Carex sartwellii	sedge	native
	Carex stricta	sedge	native
	Carex tetanica	sedge	native
	Carex viridula	sedge	native
	Cicuta maculata	forb	native
	Cirsium arvense	forb	introduced
	Cornus sericea	shrub	native
	Crataegus chrysocarpa	shrub	native
	Deschampsia cespitosa	grass	native
	Eleocharis palustris	sedge	native
	Eleocharis pauciflora	sedge	native
	Epilobium leptophyllum	forb	native
	Eriophorum angustifolium	sedge	native
	Eupatorium maculatum	forb	native
	Galium trifidum	forb	native
	Glyceria striata	grass	native
	Helianthus nuttallii	forb	native
	Juncus arcticus	forb	native

Community	Species	Physiognomy	Native/Introduced
	Juncus nodosus	forb	native
	Lycopus asper	forb	native
	Lysimachia hybrid	forb	native
	Lysimachia thyrsiflora	forb	native
	Mentha arvensis	forb	native
	Muhlenbergia mexicana	grass	native
	Muhlenbergia richardsonis	grass	native
	Parnassia palustris	forb	native
	Pedicularis lanceolata	forb	native
	Poa palustris	grass	native
	Polygonum amphibium	forb	native
	Potentilla anserina	forb	native
	Rumex occidentalis	forb	native
	Salix bebbiana	shrub	native
	Salix candida	shrub	native
	Salix petiolaris	shrub	native
	Schoenoplectus acutus	sedge	native
	Scirpus pallidus	sedge	native
	Scutellaria galericulata	forb	native
	Sonchus arvensis	forb	introduced
	Stellaria crassifolia	forb	native
	Symphyotrichum boreale	forb	native
	Symphyotrichum lanceolatum	forb	natve
	Teucrium canadense	forb	native
	Thalictrum dasycarpum	forb	native
	Triglochin maritima	forb	native
	Typha angustifolia	forb	introduced
	Typha x glauca	forb	introduced
	Utricularia intermedia	forb	native
	Viola nephrophylla	forb	native

Community	Species	Physiognomy	Native/Introduced
Floating Mat			
	Agrostis stolonifera	grass	introduced
	Apocynum cannabinum	forb	native
	Asclepias incarnata	forb	native
	Bromus ciliatus	grass	native
	Calamagrostis stricta	grass	native
	Caltha palustris	forb	native
	Carex aquatilis	sedge	native
	Carex buxbaumii	sedge	native
	Carex emoryi	sedge	native
	Carex hystericina	sedge	native
	Carex lanuginosa	sedge	native
	Carex prairea	sedge	native
	Carex sartwellii	sedge	native
	Carex sterilis	sedge	native
	Carex tetanica	sedge	native
	Carex viridula	sedge	native
	Deschampsia cespitosa	grass	native
	Eleocharis palustris	sedge	native
	Eleocharis pauciflora	sedge	native
	Elymus trachycaulus	grass	native
	Equisetum arvense	fern	native
	Eriophorum angustifolium	sedge	native
	Eupatorium maculatum	forb	native
	Euthamia graminifolia	forb	native
	Fragaria virginiana	forb	native
	Fraxinus pennsylvanica	tree	native
	Glyceria striata	grass	native
	Helianthus maximiliana	forb	native
	Helianthus nuttallii	forb	native

Community	Species	Physiognomy	Native/Introduced
	Hordeum jubatum	grass	native
	Hypoxis hirsuta	forb	native
	Juncus arcticus	forb	native
	Juncus brevicaudatus	forb	native
	Juncus dudleyi	forb	native
	Juncus nodosus	forb	native
	Liatris ligulistylis	forb	native
	Lobelia kalmii	forb	native
	Lycopus americanus	forb	native
	Lycopus asper	forb	native
	Muhlenbergia mexicana	grass	native
	Muhlenbergia richardsonis	grass	native
	Parnassia palustris	forb	native
	Pedicularis lanceolata	forb	native
	Platanthera aquilonis	forb	native
	Potentilla anserina	forb	native
	Rhynchospora capillacea	sedge	native
	Rudbeckia hirta	forb	native
	Salix bebbiana	shrub	native
	Salix candida	shrub	native
	Salix petiolaris	shrub	native
	Schoenoplectus acutus	sedge	native
	Scirpus pallidus	sedge	native
	Scutellaria galericulata	forb	native
	Senecio pseudaureus	forb	native
	Solidago canadensis	forb	native
	Solidago missouriensis	forb	native
	Sonchus arvensis	forb	introduced
	Symphyotrichum boreale	forb	native
	Symphyotrichum lanceolatum	forb	native

Community	Species	Physiognomy	Native/Introduced
	Symphyotrichum novae-angliae	forb	native
	Teucrium canadense	forb	native
	Thalictrum dasycarpum	forb	native
	Triglochin maritima	forb	native
	Triglochin palustris	forb	native
	Typha x glauca	forb	introduced
	Vicia americana	forb	native
	Viola nephrophylla	forb	native
	Zizia aurea	forb	native
Herbaceous			
	Agrostis stolonifera	grass	introduced
	Andropogon gerardii	grass	native
	Anemone canadensis	forb	native
	Apocynum cannabinum	forb	native
	Asclepias incarnata	forb	native
	Asclepias syriaca	forb	native
	Bromus ciliatus	grass	native
	Bromus inermis	grass	introduced
	Calamagrostis stricta	grass	native
	Caltha palustris	forb	native
	Carex crawei	sedge	native
	Carex interior	sedge	native
	Carex lanuginosa	sedge	native
	Carex praegracilis	sedge	native
	Carex prairea	sedge	native
	Carex sartwellii	sedge	native
	Carex tetanica	sedge	native
	Cicuta maculata	forb	native
	Cirsium arvense	forb	introduced
	Cirsium flodmanii	forb	native

Community	Species	Physiognomy	Native/Introduced
	Cirsium vulgare	forb	introduced
	Cypripedium candidum	forb	native
	Cypripedium parviflorum	forb	native
	Deschampsia cespitosa	grass	native
	Eleocharis pauciflora	sedge	native
	Elymus trachycaulus	grass	native
	Epilobium ciliatum	forb	native
	Epilobium leptophyllum	forb	native
	Equisetum arvense	fern	native
	Equisetum laevigatum	fern	native
	Eriophorum angustifolium	sedge	native
	Eupatorium maculatum	forb	native
	Euphorbia esula	forb	introduced
	Euthamia graminifolia	forb	native
	Fragaria virginiana	forb	native
	Galium boreale	forb	native
	Glyceria striata	grass	native
	Glycorrhiza lepidota	forb	native
	Helianthus maximiliana	forb	native
	Helianthus nuttallii	forb	native
	Hierochloe odorata	grass	native
	Hordeum jubatum	grass	native
	Hypoxis hirsuta	forb	native
	Juncus arcticus	forb	native
	Juncus dudleyi	forb	native
	Juncus interior	forb	native
	Liatris ligulistylis	forb	native
	Lilium philadelphicum	forb	native
	Lobelia spicata	forb	native
	Lycopus americanus	forb	native

Community	Species	Physiognomy	Native/Introduced
	Lycopus asper	forb	native
	Lysimachia ciliata	forb	native
	Muhlenbergia mexicana	grass	native
	Muhlenbergia richardsonis	grass	native
	Parnassia palustris	forb	native
	Pascopyrum smithii	grass	native
	Phleum pretense	grass	introduced
	Poa palustris	grass	native
	Poa pratensis	grass	introduced
	Polygala senega	forb	native
	Polygonum amphibium	forb	native
	Polygonum coccineum	forb	native
	Potentilla anserina	forb	native
	Prenanthes racemosa	forb	native
	Ranunculus cymbalaria	forb	native
	Ranunculus macounii	forb	native
	Rosa woodsii	shrub	native
	Rudbeckia hirta	forb	native
	Rumex occidentalis	forb	native
	Schizachyrium scoparium	grass	native
	Scirpus pallidus	sedge	native
	Senecio pseudaureus	forb	native
	Sisyrinchium campestre	forb	native
	Solidago canadensis	forb	native
	Sonchus arvensis	forb	introduced
	Spartina pectinata	grass	native
	Stachys palustris	forb	native
	Symphyotrichum falcatum	forb	native
	Symphyotrichum lanceolatum	forb	native
	Taraxacum officinale	forb	introduced

Community	Species	Physiognomy	Native/Introduced
	Teucrium canadense	forb	native
	Thalictrum dasycarpum	forb	native
	Triglochin maritima	forb	native
	Veronia fasciculata	forb	native
	Vicia americana	forb	native
	Zigadenus elegans	forb	native
	Zizia aptera	forb	native
	Zizia aurea	forb	native
Tree			
	Acer negundo	tree	native
	Agrimonia striata	forb	native
	Agrostis stolonifera	grass	introduced
	Amelanchier alnifolia	shrub	native
	Anemone canadensis	forb	native
	Apocynum cannabinum	forb	native
	Asclepias incarnata	forb	native
	Bidens frondosa	forb	native
	Bromus ciliatus	grass	native
	Bromus latiglumis	grass	native
	Calamagrostis canadensis	grass	native
	Calamagrostis stricta	grass	native
	Caltha palustris	forb	native
	Carex aquatilis	sedge	native
	Carex brevior	sedge	native
	Carex granularis	sedge	native
	Carex hystericina	sedge	native
	Carex interior	sedge	native
	Carex lanuginosa	sedge	native
	Carex sartwellii	sedge	native
	Carex sterilis	sedge	native

Community	Species	Physiognomy	Native/Introduced
	Carex tetanica	sedge	native
	Carex utriculata	sedge	native
	Cicuta maculata	forb	native
	Cirsium arvense	forb	introduced
	Cornus sericea	shrub	native
	Deschampsia cespitosa	grass	native
	Eleocharis palustris	sedge	native
	Epilobium ciliatum	forb	native
	Epilobium leptophyllum	forb	native
	Equisetum arvense	fern	native
	Equisetum laevigatum	fern	native
	Erigeron philadelphicus	forb	native
	Eupatorium maculatum	forb	native
	Euphorbia esula	forb	introduced
	Euthamia graminifolia	forb	native
	Fragaria virginiana	forb	native
	Fraxinus pennsylvanica	tree	native
	Galium aparine	forb	native
	Galium boreale	forb	native
	Geum allepicum	forb	native
	Glyceria striata	grass	native
	Glycorrhiza lepidota	forb	native
	Helianthus nuttallii	forb	native
	Juncus interior	forb	native
	Lathyrus ochroleucus	forb	native
	Lycopus americanus	forb	native
	Lycopus asper	forb	native
	Lysimachia ciliata	forb	native
	Lysimachia thyrsiflora	forb	native
	Mentha arvensis	forb	native

Community	Species	Physiognomy	Native/Introduced
	Muhlenbergia mexicana	grass	native
	Muhlenbergia racemosa	grass	native
	Parthenocissus quinquefolia	forb	native
	Pedicularis lanceolata	forb	native
	Phalaris arundinacea	grass	native
	Phleum pratense	grass	introduced
	Platanthera aquilonis	forb	native
	Poa palustris	grass	native
	Poa pratensis	grass	introduced
	Potentilla anserina	forb	native
	Prunis virginiana	shrub	native
	Ranunculus macounii	forb	native
	Ribes americanum	shrub	native
	Rosa woodsii	shrub	native
	Rubus ideaus	shrub	native
	Salix bebbiana	shrub	native
	Salix candida	shrub	native
	Salix petiolaris	shrub	native
	Scirpus pallidus	sedge	native
	Scutellaria galericulata	forb	native
	Senecio pseudaureus	forb	native
	Sisyrinchium campestre	forb	native
	Solidago canadensis	forb	native
	Sonchus arvensis	forb	introduced
	Spartina pectinata	grass	native
	Stellaria crassifolia	forb	native
	Symphoricarpos occidentalis	shrub	native
	Symphyotrichum lanceolatum	forb	native
	Symphyotrichum puniceum	forb	native
	Taraxacum officinale	forb	introduced

Community	Species	Physiognomy	Native/Introduced
	Teucrium canadense	forb	native
	Thalictrum dasycarpum	forb	native
	Thalictrum venulosum	forb	native
	Toxicodendron radicans	forb	native
	Typha x glauca	forb	introduced
	Vicia americana	forb	native
	Viola nephrophylla	forb	native
	Zizia aurea	forb	native
Wet Meadow			
	Acer negundo	tree	native
	Agrostis stolonifera	grass	introduced
	Andropogon gerardii	grass	native
	Apocynum cannabinum	forb	native
	Artimesia absinthium	forb	introduced
	Asclepias syriaca	forb	native
	Bromus inermis	grass	introduced
	Calamagrostis stricta	grass	native
	Carex aquatilis	sedge	native
	Carex crawei	sedge	native
	Carex interior	sedge	native
	Carex lanuginosa	sedge	native
	Carex praegracilis	sedge	native
	Carex tetanica	sedge	native
	Cicuta maculata	forb	native
	Cirsium arvense	forb	introduced
	Cirsium vulgare	forb	introduced
	Cornus sericea	shrub	native
	Deschampsia cespitosa	grass	native
	Eleocharis palustris	sedge	native
	Eleocharis pauciflora	sedge	native

Community	Species	Physiognomy	Native/Introduced
	Elymus repens	grass	introduced
	Elymus trachycaulus	grass	native
	Epilobium leptophyllum	forb	native
	Equisetum laevigatum	fern	native
	Eupatorium maculatum	forb	native
	Euphorbia esula	forb	introduced
	Euthamia graminifolia	forb	native
	Fragaria virginiana	forb	native
	Glyceria striata	grass	native
	Glycorrhiza lepidota	forb	native
	Helianthus maximiliana	forb	native
	Helianthus nuttallii	forb	native
	Hierochloe odorata	grass	native
	Hordeum jubatum	grass	native
	Hypoxis hirsuta	forb	native
	Juncus arcticus	forb	native
	Juncus dudleyi	forb	native
	Juncus interior	forb	native
	Liatris ligulistylis	forb	native
	Lobelia spicata	forb	native
	Lycopus asper	forb	native
	Lysimachia ciliata	forb	native
	Mentha arvensis	forb	native
	Muhlenbergia richardsonis	grass	native
	Phleum pratense	grass	introduced
	Poa pratensis	grass	introduced
	Potentilla anserina	forb	native
	Ranunculus cymbalaria	forb	native
	Ranunculus pensylvanicus	forb	native
	Rosa woodsii	shrub	native

Community	Species	Physiognomy	Native/Introduced
	Rudbeckia hirta	forb	native
	Schizachyrium scoparium	grass	native
	Scirpus pallidus	sedge	native
	Senecio pseudaureus	forb	native
	Sisyrinchium campestre	forb	native
	Solidago canadensis	forb	native
	Sonchus arvensis	forb	introduced
	Spartina pectinata	grass	native
	Symphoricarpos occidentalis	shrub	native
	Symphyotrichum falcatum	forb	native
	Symphyotrichum lanceolatum	forb	native
	Teucrium canadense	forb	native
	Thalictrum dasycarpum	forb	native
	Triglochin palustris	forb	native
	Typha angustifolia	forb	introduced
	Typha x glauca	forb	introduced
	Viola nephrophylla	forb	native
	Zigadenus elegans	forb	native
	Zizia aptera	forb	native
	Zizia aurea	forb	native

APPENDIX C. SPECIES CORRELATIONS (Pearson │r│> 0.4) WITH NONMETRIC MULTIDIMENSIONAL SCALING (NMS) AXES

Axis	Species	Species Correlation
Axis 1		
	Agrostis stolonifera	0.70
	Andropogon gerardii	0.49
	Anemone canadensis	0.54
	Bromus inermis	0.56
	Calamagrostis stricta	0.86
	Carex buxbaumii	-0.41
	Carex crawei	0.53
	Carex emoryi	-0.46
	Carex lanuginosa	0.48
	Carex praegracilis	0.44
	Carex prairea	-0.45
	Carex stricta	-0.42
	Carex viridula	-0.48
	Cirsium arvense	0.48
	Cirsium vulgare	0.50
	Crataegus chrysocarpa	-0.42
	Cypripedium parviflorum	0.46
	Eleocharis pauciflora	-0.44
	Elymus trachycalulus	0.41
	Equisetum laevigatum	0.87
	Euphorbia esula	0.58
	Euthamia graminifolia	0.46
	Fragaria virginiana	0.51
	Galium boreale	0.41
	Glyceria striata	0.59
	Glycorrhiza lepidota	0.43

Axis	Species	Species Correlation
	Helianthus nuttallii	0.62
	Hierochloe odorata	0.51
	Juncus brevicaudatus	-0.44
	Juncus interior	0.48
	Juncus nodosus	-0.42
	Lobelia kalmii	-0.44
	Lysimachia ciliata	0.63
	Lysimachia hybrida	-0.42
	Lysimachia thyrsiflora	-0.42
	Muhlenbergia mexicana	-0.51
	Pdicularis lanceolata	-0.67
	Phleum pratense	0.44
	Poa pratensis	0.71
	Potentilla anserina	0.67
	Ranunculus cymbalaria	0.44
	Ranunculus macounii	0.53
	Rosa woodsii	0.52
	Salix candida	-0.44
	Schizachyrium scoparium	-0.59
	Schoenoplectus acutus	0.40
	Scirpus pallidus	0.41
	Senecio pseudaureus	0.50
	Sisyrinchium campestre	0.59
	Solidago canadensis	0.59
	Sonchus arvensis	0.66
	Spartina pectinata	0.59
	Symphyotrichum boreale	-0.63
	Symphyotrichum falcatum	0.65
	Symphyotrichum lanceolatum	0.51
	Taraxacum officinale	0.54

Axis	Species	Species Correlation
	Thalictrum dasycarpum	0.50
	Typha angustifolia	-0.41
	Typha x glauca	-0.48
	Utricularia intermedia	-0.42
	Viola nephrophylla	-0.56
	Zizia aurea	0.62
Axis 2		
	Carex aquatilis	-0.41
	Carex buxbaumii	-0.58
	Carex emoryi	-0.67
	Carex sartwellii	-0.41
	Carex sterilis	-0.54
	Carex viridula	-0.45
	Epilobium leptophyllum	0.64
	Eriophorum angustifolium	-0.52
	Galium trifidum	0.71
	Hypoxis hirsuta	-0.54
	Juncus brevicaudatus	-0.68
	Liatris ligulistylis	-0.48
	Lobelia kalmii	-0.68
	Lycopus americanus	-0.41
	Lysimachia thyrsiflora	0.62
	Mentha arvense	0.77
	Muhlenbergia mexicana	-0.71
	Parnassia palustris	-0.78
	Pedicularis lanceolata	-0.60
	Rhynchospora capillacea	-0.43
	Rudbeckia hirta	-0.45
	Salix candida	-0.55
	Schoenoplectus acutus	-0.40

Axis	Species	Species Correlation
	Solidago missouriensis	-0.48
	Symphyotrichum boreale	-0.67
	Symphyotrichum novae-angliae	-0.48
	Triglochin maritima	-0.66
	Triglochin palustris	-0.43
	Typha x glauca	0.73
Axis 3		
	Acer negundo	-0.52
	Agrimonia striata	-0.41
	Amelanchier alnifolia	-0.59
	Andropogon gerardii	0.50
	Asclepias incarnata	-0.51
	Asclepias syriaca	0.51
	Bromus ciliatus	-0.50
	Bromus latiglumis	-0.41
	Calamagrostis canadensis	-0.66
	Caltha palustris	-0.69
	Carex brevior	-0.77
	Carex granularis	-0.59
	Carex praegracilis	-0.41
	Carex utriculata	0.44
	Cornus sericea	-0.59
	Deschampsia cespitosa	0.54
	Eleocharis pauciflora	0.46
	Elymus trachycaulus	0.46
	Equisetum arvense	-0.61
	Erigeron philadelphicus	-0.72
	Fraxinus pennsylvanica	-0.63
	Galium aparine	-0.59
	Geum aleppicum	-0.41

Axis	Species	Species Correlation
	Helianthus nuttallii	0.50
	Hordeum jubatum	0.55
	Juncus articus	0.49
	Mentha arvense	-0.45
	Muhlenbergia richardsonis	0.43
	Parthenocissus quinquefolia	-0.65
	Plantheria aquilonis	-0.46
	Prunis virginiana	-0.50
	Ranunculus cymbalaria	0.44
	Ribes americanum	-0.72
	Rubus idaeus	-0.41
	Salix bebbiana	-0.61
	Salix petiolaris	-0.76
	Sonchus arvensis	0.42
	Symphyotrichum falcatum	0.46
	Symphyotrichum puniceum	-0.59
	Thalictrum dasycarpum	-0.42
	Thalictrum venulosum	-0.86
	Toxicodendron radicans	-0.58
	Triglochin palustris	0.42
	Zizia aptera	0.51