ASSESSMENT OF PRAIRIE POTHOLE CONDITIONS AND PLANT COMMUNITY COMPOSITION ON FWS FEE-TITLE LANDS

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Assessment of Prairie Pothole conditions and plant community composition on FWS fee-title lands

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ABSTRACT

Conditions of wetlands in the Prairie Pothole Region have been severely degraded due to anthropogenic disturbances, such as cultivation and climate change. To maintain or restore the diversity and integrity of these ecosystems we must first understand what condition they are in and what current factors are driving wetland conditions on a region-wide scale. This study aimed to assess wetland conditions and determine what the major plant community drivers were on FWS fee-title lands. Assessments showed wetlands in native grassland are in better condition than those in reseeded grasslands and seasonal wetlands are in better condition than temporary wetlands. It was clear plant communities are being largely driven by the cover of invasive species within each given wetland zone. Differences in wetland conditions and invasive versus native species cover are likely the result of past and present disturbance on FWS fee-title lands.

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LIST OF ABBREVIATIONS

ANOVA ....Analysis of Variance

C-value ....Coefficient of Conservatism

FQA..Floristic Quality Assessments

FQI ....Floristic Quality Indices

FWS ....United States Fish and Wildlife Service

GLM...Generalized Linear Model

GIS ....Geographic Information System

GPS ....Global Positioning System

HGM ....Hydrogeomorphic

IBI ....Index of Biological Integrity

IPCI..Index of Plant Community Integrity

LWCAM ....Landscape Wetland Condition Analysis Model

MRPP...Multi-response Permutation Procedure

NDRAM...North Dakota Rapid Assessment Method

NMS...Nonmetric Multi-dimensional Scaling

NPAM..Native Prairie Adaptive Management

PPR ....Prairie Pothole Region

RAM ....Rapid Assessment Method

WMD ....Wetland Management District

1. INTRODUCTION

1.1. Wetlands Overview

The importance of wetlands is generally not well understood by much of the public, but those who understand the ecological processes of wetlands often consider them some of the most productive ecosystems. For example, comprising only 4% of terrestrial land cover they account for 33% of the total soil organic matter (Whittaker and Likens 1973; Eswaran et al. 1993; Euliss et al. 2006). The definition of a wetland is variable, just like wetlands themselves, and often differs depending on the type of wetland, region, and the party defining it. One of the most commonly accepted definitions in the U.S. is that which is used by the U.S. Army Corps of Engineers and the U.S. Environmental Protection Agency for regulatory purposes regarding the Clean Water Act since the 1970s (USCOE 1987). Here wetlands are defined as "areas that are inundated or saturated by surface or ground water at a frequency and duration sufficient to support, and that under normal circumstances do support, a prevalence of vegetation typically adapted for life in saturated soil conditions". Another commonly accepted definition is that of the U.S. Fish and Wildlife Service (FWS) which states "Wetlands are lands transitional between terrestrial and aquatic systems where the water table is usually at or near the surface or the land is covered by shallow water" (Cowardin et al. 1979). This nonregulatory definition also states three requirements to be considered a wetland: hydrophytic vegetation must be present, soils present must be predominantly undrained hydric soils, and it must be saturated with water or covered with shallow water during at least some point during the year. Both definitions are widely accepted and either can be used for the purpose of research in the Prairie Pothole Region (PPR).

1.1.1. PPR Characteristics

The PPR currently covers over 770,000 square kilometers through parts of Montana, North Dakota, South Dakota, Minnesota, Iowa, Alberta, Saskatchewan, and Manitoba (Dahl 2014; Gleason and Tangen 2014). Its formation started roughly 12,000 years ago during the Pleistocene glacial retreat (Johnson et al. 2008). The PPR experiences variable temperatures, precipitation, evapotranspiration rates, topography, and land-uses across latitudinal and longitudinal gradients (Rothrock 1943; Bluemle 2000; Millet et al. 2009). This region can be separated into smaller classes of ecoregions, such as the Prairie Coteau, Missouri Coteau, and Glaciated Plains (Bluemle 2000). These ecoregions differ in climate, land-use, topography, and hydrology. The Glaciated Plains on the eastern side is characterized by a gently sloping, rolling landscape with the Lake Agassiz Plain and Turtle Mountains ecoregions often considered to be within it, while the Missouri Coteau and Prairie Coteau to the west and south respectively are generally hummocky plains of glacial sediment. Each ecoregion also contains different concentrations of potholes in addition to differences in physical and climatic characteristics, with the Missouri Coteau having the highest pothole wetland concentration.

The PPR is characterized as having long cold winters and short hot summers (Johnson et al. 2005). It follows a temperature gradient running North to South where annual mean temperature ranges from 1 degree Celsius in its northernmost reaches of Canada to 10 degrees Celsius in its southernmost reaches of Iowa and Nebraska (Millett et al. 2009). The highest observed temperature variations over recent decades have been for winter temperatures while summer temperatures have seen less change, and daily minimum temperatures have increased more than daily maximum temperature averages. The precipitation gradient for the most part goes west to east and is correlated to the ecoregions. The average annual precipitation for the northwesternmost extent of the study area is less than 400 mm per year while the southeasternmost extent has an average of over 600 mm per year.

1.1.2. PPR Wetland Characteristics

After the last glacial retreat, glacial ice incorporated with glacial till melted, which formed many closed depressions with low-permeability soil that collected water to develop into the prairie potholes we see today (Johnson et al. 2008). A substantial portion of these potholes have been lost to anthropogenic disturbance such as agricultural land use (Pennock et al. 2010). However, in the PPR, more than 2.6 million potholes remain comprising roughly 26,000 square kilometers of the region. Prairie potholes are distinguished by unique hydrologic, biotic, chemical, and physical characteristics.

The depressional wetlands of the PPR are a part of a generally flat to hummocky or undulating landscape (Hayashi et al. 2016). The topography, climate, soil permeability, and land-use all affect the hydrology of potholes (Winter and Rosenberry 1995; Euliss and Mushet 1996; Werner et al. 2013; Hayashi et al. 2016). Evapotranspiration rates exceed precipitation rates in the PPR, meaning potholes are heavily dependent on lateral water inputs from the upland, such as runoff from snowmelt or heavy rains (Hyashi et al. 2016). This can vary due to year to year or decadal changes in precipitation. Lateral water inputs and pond permanence also often vary depending on relative position within the landscape. Potholes relatively higher topographically tend to contribute the most to groundwater and lower wetlands, and are termed recharge wetlands, where those in a topographically low position, known as discharge wetlands, receive groundwater, and intermediate wetlands are known as flow-through wetlands (Lissey 1971; Euliss et al. 2004). Surface-water connectivity also varies with topography and is important for water flowing through wetland complexes (Shaw et al. 2012; Euliss et al. 2004). Surface-water flows from wetlands at higher positions within the landscape to those at lower positions, similarly to groundwater. This causes wetlands in a higher position to tend to be smaller and less permanently ponded and lower positioned wetlands to be larger and more permanently ponded with higher salinity (Hayashi et al. 2016).

These potholes are also characterized by having an underlying layer of clay-rich till with a high water holding capacity, allowing them to be highly saturated up to 1 meter above the water table (Hayashi et al. 2016). The combination of these topographical, climatic, and soil permeability features cause pothole water level patterns to have high seasonality, unlike wetlands in more humid environments able to sustain steadier water levels due to sufficient water inflow and outflow throughout the year. Water levels tend to be highest after the spring snowmelt, with a gradual decline through the summer and fall. By late summer or fall it is common for even the central area of the pothole to become dry. Even during and after this dry period, hydrological processes continue due to soil water and groundwater processes below the surface. These processes are large driving factors in determining what vegetation is present in and around wetlands (Stewart and Kantrud 1971; Euliss et al. 2004; Hayashi et al. 2016).

Palustrine emergent wetlands are by far the most numerous and widely distributed wetland type in the PPR (Cowardin et al. 1979). Palustrine emergent wetlands are nontidal wetlands dominated by persistent emergent vegetation usually small in size (<20 acres), have shallow water depths often less than 2 meters, have no wave formed shoreline, and contain salts not derived from the ocean. Potholes are typically classified based on water permanency and vegetation zonation (Stewart and Kantrud 1971; Niemuth et al. 2010). Pothole classifications include ephemeral, temporary, seasonal, semi-permanent, and permanent wetlands. These types of wetlands often differ not only in vegetation zones and water permanence, but also abundance, size, landscape position, cover type, and physiochemical properties.

Temporarily and seasonally ponded potholes are the most commonly occurring pothole types. About 90 percent of remaining potholes are classified as temporarily or seasonally ponded with 2 vegetation zones for temporarily ponded wetlands and 3 vegetation zones for seasonally ponded wetlands (Steward and Kantrud 1971). These two wetland classifications have an exterior low-prairie zone and an interior wet-meadow zone, while seasonally ponded wetlands also include a central shallow marsh zone. Other pothole classifications can also include deep-marsh, permanent-open-water, intermittent-alkali, and fen zones. Van der Kamp et al. (2016) has suggested updating much of the terminology used related to wetlands, including the definition of wetlands itself, due to its reliance on vegetation and water permanence over hydric soils, but for the purpose of most research in the PPR the classification systems and definitions proposed by Stewart and Kantrud (1971) have been deemed sufficient.

1.1.3. Ecosystem Services

The value of PPR wetlands coincides with the biodiversity and ecosystem services they offer. On top of harboring many important species of plants, they also act as crucial wildlife habitat, and provide other ecosystem services such as flood mitigation, filtration of pollutants, groundwater recharge, nutrient retention, water for livestock, and recreational opportunities (Winter and Rosenberry 1995; Euliss et al. 2006; Badiou et al. 2011; Gleason et al. 2011; Hayashi et al. 2016). Flood mitigation and groundwater recharge are heavily dependent on the complex and variable surface and subsurface hydrological processes within individual wetlands and wetland complexes (Hayashi et al. 2016). Key characteristics correlated to these, and other wetland ecosystem services, such as biodiversity, plant community structure, hydrology, and soils can be greatly affected by changes in land-use and climate (Euliss and Mushet 1996; Gleason and Euliss 1998; DeKeyser et al. 2003; Gleason et al. 2003; Balas et al. 2012; Werner et al. 2013). Changes in these key aspects often reflect changes in wetland condition as well. The extent to which potholes provide potential ecosystem services is often dependent on the condition of the wetland. Possibly the most important values in need of the most focus are those of biodiversity and wildlife habitat (Dixon et al. 2019). PPR wetlands provide important habitat for many species of migratory waterfowl, non-game birds, mammals, reptiles, amphibians, honeybees, and some other native pollinators (Balas et al. 2012; Smart et al. 2017; Dixon et al. 2019).

1.2. PPR Wetland Disturbance

There is a wide variety of disturbances occurring throughout the PPR ranging from historically naturally occurring disturbance (e.g. fire and grazing) to anthropogenically related disturbances (e.g. cultivation, drainage, climate change, urbanization, ditching, sedimentation, and chemical runoff) (Kantrud et al. 1989; Gleason and Euliss 1998; Galatowitsch et al. 2000; Steen et al. 2016). The manner and severity for which potholes are affected by disturbance depends on the type of disturbance. For this context disturbance can be defined as "any series of events that disrupt ecosystem, community, and population structure and alters the physical environment by natural or unnatural means" (Smith 2011).

1.2.1. Land Use Disturbance

Agricultural expansion, or the conversion of native prairie to croplands, has been one of the most significant driving factors affecting PPR wetlands in the last 150 years (McKenna et al. 2019). Up to 90% of temporary and seasonal wetlands have been lost due to this (Knutsen and Euliss 2001). This has also led some areas of the PPR to have an even higher percentage loss of native prairie uplands (Samson and Knopf 1994). Cultivation accompanying this extensive conversion to agricultural land has led to increased soil erosion, sedimentation, and overall wetland degradation (Kantrud et al. 1989). While most of the cultivated land has been in the uplands, the wet prairie and sedge meadow zones of potholes are also regularly cultivated (Galatowitsch and van der Valk 1996b). These zones experience less flooding and have more desirable soils for high crop yields than more interior wetland zones meaning they are cultivated more often than not in agricultural areas.

Agricultural expansion is still ongoing and expected to continue (Johnston 2013; Johnston and McIntyre 2019). This has led to a decrease in wetland size and density in recent years with decreases expected to continue with further agricultural expansion. Johnston and McIntyre (2019) found that while cumulative wetland area decreased by 25%, density decreased by 16%, and average size decreased by over 10%, there was no significant loss in structural connectivity. As climate shifts and agricultural expansion continue there will likely be a tipping point for which structural connectivity will be greatly affected, but it is uncertain when the tipping point will be reached. The other major expansion threatening PPR wetlands is that of urbanization (Galatowitsch et al. 2000). The PPR is not a very densely populated area, but as populations continue to move from more rural to urban areas, and cause those urban areas to expand outwards, wetland losses will follow.

Alterations to hydrology (drainage, ditching, and runoff alterations) are the other most severe direct anthropogenic disturbances negatively affecting PPR wetlands today (Galatowitsch et al. 2000; McKenna et al. 2019). McKenna et al. (2019) found when upland wetlands are drained into terminal wetlands, the terminal wetlands reach their spill point 4 years early on average. This led to 10 times the normal amount of water spilling into local stream networks and consequently greatly increasing flooding risks. Galatowitsch et al. (2000) also found hydrological alterations to have extensive impacts on wetland vegetation. Alterations often increased the abundance of less desirable species, changed life history guilds, and decreased overall species richness. Hydrology and plant community composition are two of the most important factors for wetland ecosystem services. Therefore, altering them in a negative way will decrease the extent to which wetlands provide desirable ecosystem services.

1.2.2. Climate Change in the PPR

Prairie Pothole wetlands are an increasingly studied subject when it comes to climate change because of their vulnerability (Steen et al. 2014; Steen et al. 2016; Rashford et al. 2016). They are particularly susceptible to climatic variation due to their small size and already relatively dry regional climate (Steen et al. 2014). Climate models for the region show an increase in temperature of 2-4 degrees Celsius by the end of the century, accompanied by slight to no further increase in precipitation and higher rates of evapotranspiration, ultimately leading to fairly dramatic changes in wetland hydroperiods, vegetative condition, water depth, and basin size (Steen et al. 2014; Johnson et al. 2016). While a climate driven change in prairie pothole wetlands depends largely on alteration of natural processes, human responses to those changes in and around the wetlands will be a major factor as well (Rashford et al. 2016).

The primary way climate change directly and indirectly affects ecological components of prairie pothole wetlands is through the alteration of their hydrology, which happens to be the most important factor in controlling key wetland processes and services (Johnson et al. 2010). Prairie pothole wetlands generally go through a series of hydrological cycles on an annual and interannual basis (Euliss et al. 2004; Johnson et al. 2010). These cycles include periods of flooding and higher water levels after snow melt and in times of high precipitation alternating with levels of little to no surface water in times of drought and high temperatures. These stages are often referred to as deluge and drought phases (Euliss et al 2004; Johnson et al. 2016). Even minor changes of precipitation, evapotranspiration, and temperature can greatly affect these cycles.

While much of the PPR has been experiencing increases in precipitation over the last few decades, it has been accompanied by an increase temperature leading to higher evapotranspiration rates within wetlands (Winter 2000; Johnson et al. 2016). This means a reduction of surface water area and reduced summer soil moisture for potholes. Another likely consequence would be increased demands on groundwater, which could result in earlier drying of wetlands. Other important ways prairie pothole wetland water levels are affected by increased temperature include a longer length of the ice-free season and change in the sublimation rate of the snowpack affecting spring runoff (Johnson et al. 2016). All of these aspects can greatly affect the start time, end time, and duration of major hydrological cycles pertaining to prairie pothole wetlands on both short-term and long-term scales.

Changes in hydrology of prairie pothole wetlands has been shown to have major effects on vegetation cover cycles in and around the wetlands (Euliss et al. 2004; Johnson et al. 2016). These vegetation cover cycles are largely driven by the dry and wet extremes of the climate (i.e. drought and deluge phases). They are also largely dictated by groundwater discharge and recharge (Euliss et al. 2004). The seasonal, annual, and interannual fluctuations in vegetation cover cycles in response to these wetland processes is often referred to as the wetland continuum. The speed and completeness of these vegetation cover cycles is mostly responsible for the productivity, structure, composition, and biodiversity of prairie pothole wetland vegetation (Euliss et al. 2004; Johnson et al. 2005; Groffman et al. 2006). For proper speed and completion of these vegetation cycles the wetland must go through high water deluge phases where there is little emergent plant cover and few nutrients in detritus, along with persistent low water drought phases with high emergent plant cover and high nutrient plant sequestration (Johnson et al. 2016). Completely going through both extremes during annual and interannual weather cycles allows for higher plant population turnover and the maintenance of high primary productivity, secondary productivity, and biological diversity. In vegetation cycle terms these extremes can be classified as wet-marsh and dry-marsh stages, with an intermittent, and most productive, hemi-marsh stage. In order to complete a proper vegetation cycle two key thresholds, such as switching between stages, must be completed (Johnson et al. 2005). Switching stages allows for certain species to germinate properly and replace others killed off by the switch (Poiani et al. 1995; Johnson et al. 2016). Changes in climate could cause these wetlands to get stuck in one stage for far too long leading to a much less productive and biodiverse ecosystem.

Another undesirable consequence of climate change involves dominance of species more adaptable to change and tolerant to less than ideal wetland conditions (Larkin et al. 2012; Johnson 2019). This could potentially mean an abundance of invasive species, such as *Typha x glauca* (hybrid cattail), consequently leading to less biodiversity and ecological productivity. Fewer overall numbers of wetlands and an overall drier climate will likely lead wetlands to be more extensively covered by vegetation alone rather than a mixture of open water and vegetation (Johnson et al. 2010). Of all impacts climate change and increasing temperature may have on prairie pothole wetland vegetation, the largest will likely be historically productive areas becoming persistently unproductive due to lower vegetative cycling rates and longer periods in the dry marsh stage (Johnson et al. 2016). It is also possible for many potholes to actually show signs of improved biological production from modest prolonged drying periods. Although, they most likely will eventually experience a change to a persistently unproductive state.

Climate change will likely indirectly affect many wildlife species through the alteration of hydrology and vegetation (Steen et al. 2014; Steen et al. 2016). In terms of vulnerability, migratory water birds that use prairie potholes as a stop along their migration, and more importantly as breeding grounds, appear to be the most common topic of concern. Amphibians, invertebrates, raptors, passerines, fish, and other wildlife can be very vulnerable to these changes as well. Prairie pothole wetlands are so crucial to migratory waterfowl because a majority of the total population of North American waterfowl hatch in the PPR. They also provide major food sources necessary to sustain many breeding and migrant waterfowl. There is high variability in the extent of vulnerability of waterfowl species to climate change depending on their preferences in wetland size, permanence, and vegetative cover for habitat selection (Steen et al. 2014). Species associated with deeper water are likely to see smaller negative impacts than those who require shallow and often temporary wetlands. This is due to a lower magnitude of impact climate change will have on deep water wetlands (Steen 2016). However, even species with a preference for deeper and more permanently ponded wetlands often still rely on smaller temporary and seasonal wetlands to some extent. In addition, alterations to hydrology of smaller temporary and seasonal wetlands will affect groundwater recharge and discharge cycles, ultimately leading to changes in deeper more permanently ponded wetland hydrology (Euliss et al. 2004). Overall climate change will likely cause large decreases in habitat for many species of water birds and other wetland dependent species (Steen et al. 2014). All of these factors, along with the projected loss of the total range of suitable prairie potholes, have the possibility to cause large decreases in populations of waterfowl in the decades to come, especially in those species most vulnerable (Steen et al. 2016).

The loss of waterfowl habitat, total population, and distribution could have negative side effects for an important ecosystem service for many in the form of recreational hunting (Johnson 2019). Hunting of waterfowl is a common recreational activity in the PPR. It provides a large revenue which often partially goes back into sustaining waterfowl species and their habitat (Rubio-Cisneros et al. 2014; Dixon et al. 2019). Waterfowl hunting also provides an additional ecosystem service in the form of a source a food for many people. Other recreational benefits of prairie potholes threatened by climate change include bird watching, fishing, canoeing, and kayaking (Johnson 2019).

The effects of climate change on potholes can also be felt by many farmers and ranchers (Johnson 2019). Farmers commonly have a negative view of potholes as being impediments for their large farm equipment and constraints to their total cropland. These potholes affect them in other ways as well. As climate changes alter potholes, economic incentives will cause farmers to have to begin to consider land-use change such as conversion of nearby pastureland to crop fields to account for losses (Rashford et al. 2016). Adversely, the changes made by farmers in the surrounding area could have an even greater impact on potholes. Potholes are often viewed positively by ranchers (Johnson 2019). The potholes provide a water source to their livestock at little to no expense. Shallow wetlands also provide high yields of forage for livestock later in the summer after they have dried out. Dense wetland vegetation can even provide shelter to livestock during inclement weather. However, as climate change alters the water and vegetation dynamics of prairie potholes, ranchers could potentially lose the benefits prairie potholes are providing for their livestock.

While not considered as often as the ecological impacts, these social components will likely be affected and need to be adjusted as climate change alters prairie pothole ecosystems. It is difficult to precisely determine the overall social-ecological impact climate change will have on prairie pothole wetlands. Most models do not provide much hope for optimism, but changes to mitigate the effects of climate change and proper management could still lead to the longevity and overall resilience of these important ecosystems (Steen et al. 2014; Johnson et al. 2016; Rashford et al. 2016).

1.3. Management

The beginning of wetland management by the FWS in the PPR began with the acquisition of land dating back to 1934 when the Duck Stamp Act was passed (Dixon et al. 2019). The passing of this act led to the establishment of National Wildlife Refuges with the aim of protecting important waterfowl habitat. In the 1950s, the Small Wetlands Acquisition Program led to the creation of wetland management districts (WMDs) and the purchase of many smaller waterfowl production areas. Most of the early management of these lands was focused on wetlands and waterfowl rather than upland grasslands.

The way many of these acquired lands appear today is the result of much earlier management strategies. Early research on these lands showed waterfowl preference for tall and dense upland nesting cover (Dixon et al. 2019). This led to the widespread seeding of several introduced species such as *Medicago sativa* (alfalfa), *Melilotus officinalis* (sweet clover), and *Thinopyrum intermedium* (intermediate wheatgrass) which still often dominate certain upland landscapes today (Duebbert and Kantrud 1974). Management often left these lands idle because of assumptions that natural historic disturbances such as fire and grazing which waterfowl evolved with would be detrimental to waterfowl nesting habitat, but these management strategies actually led to the quick deterioration of these lands. Another early management strategy involved the replanting of former cropland with non-native species (Duebbert et al. 1981). This coupled with the idle grassland management strategies of the time is likely largely responsible for the spread of these non-native species to other grassland and wetland areas throughout the PPR.

By the 1990s management strategies changed to more frequently include fire and grazing, and reseeding mixtures were updated to include mostly native grasses, shrubs, and forbs (Dixon et al. 2019). Updated management and research strategies along with a growing concern over invasive cool-season grasses led the FWS to develop the Native Prairie Adaptive Management initiative (NPAM) (Grant et al. 2009; Dixon et al. 2019). NPAM continues to be used to help determine the type and frequency of management to use on a given year (e.g. rest, fire, grazing), how restoration is being affected by invasive cool-season grasses, what effects previous years of management have had, and the effectiveness of the restoration efforts. The use of NPAM has led to slight increases in native plant composition and ecological integrity of FWS fee-title native prairie uplands. The new adaptive management strategies of the FWS are continuously being updated to increase native plant diversity and allow for natural function of the prairies. While recently many efforts have been taken to restore and reconstruct grasslands to reflect conditions similar to native prairie, there has been less focus on wetlands in these areas. The actions being taken such as prescribed burning, grazing, and the reseeding of grasslands to prairie from an agricultural setting all likely have some positive impacts on wetlands in the area but are targeted more at improvement of the uplands than the wetlands. Plant communities of wetlands are distinctly different from upland prairie and therefore may require different or additional management techniques for wetland functions and plant communities to be restored on former agricultural land or maintained in native prairies. This should be a major focus for the FWS in upcoming years considering the importance of wetlands for wildlife habitat and the vast array of other ecosystem services they provide. This will likely include restoration and reseeding efforts aimed at restoring wetland functions, hydrology, and native diversity accompanied by implementation of various management regimes similar to NPAM (rest, grazing, fire).

Similar to the upland sections of FWS fee-title lands, wetlands on these lands have experienced ecological issues with the lack of natural disturbance and introduction of invasive species. The current burning and grazing patterns of these lands follows the schedule put forth using NPAM data (Dixon et al. 2019). This ultimately means wetlands on those lands are also experiencing periodic grazing and fire. While it is designed to follow a pattern for what is best for the upland areas and not necessarily what type and frequency of disturbance would be best for wetlands, it may in fact be increasing the diversity and ecological integrity of wetlands as well. However, the relationship has not been extensively assessed and even if said disturbances are proving beneficial to wetlands, a majority of wetlands on FWS fee-title land have remained without fire for many years. In the cropland dominated areas of the state, where there tend to be fewer cattle for grazing management, FWS fee-title lands often remain completely idle for many years. This idle management has led to many of these wetlands, especially those in reseeded grasslands, to become invaded and dominated by introduced species.

Many research studies have focused on the best ways to restore native plant diversity of wetlands in the PPR, and most have come to one of 4 conclusions: 1) reduce the dominance of invasive plant species (Galatowitsch et al. 1999; Bansal et al. 2019), 2) the development of vegetation zones, specifically the wet meadow zone, is key to restoring native wetland vegetation (Seabloom and van der Valk 2003a), 3) past disturbance (e.g. cultivation) was too great and restored wetland plant communities differ too greatly from native wetland plant communities to achieve restoration to reflect native conditions (Galatowitsch and van der Valk 1996a; 1996b), and 4) wetland complexes should be the focus of wetland restoration efforts rather than individual wetlands. Regardless of which conclusion is most accurate, it is necessary for wetland restorations to be assessed on a case-by-case basis because of the varying past and present disturbances, conditions, and management regimes.

Fire, grazing, and herbicide application have all proven to be moderately successful strategies for limiting the dominance of invasive species and restoring native plant diversity in both wetlands and the surrounding uplands (Bansal et al. 2019; Larson et al. 2020). Time of year, frequency, invasive species present, and continued monitoring and management are all important aspects when using any of these disturbances for management purposes. Restoration of natural hydrology is also a very important factor, but this has already been a major priority of the FWS since the time they first started attaining and managing fee-title lands (Dixon et al. 2019). Wetland restoration and management should be of the utmost importance for the FWS in coming years, but in order to determine and apply best management practices they need to know the conditions of their wetlands and what factors are most significantly affecting wetland conditions.

1.4. Condition Assessments

There has been a substantial amount of wetland destruction and degradation across the PPR, and other regions of the U.S.A., due mostly to anthropogenic disturbance (Hargiss et al. 2017; Kentula and Paulsen 2019). This had led to a need to address questions about current wetland conditions and the best ways to assess those wetlands (Hargiss et al. 2017). Wetland condition assessments are common methods used to see how anthropogenic disturbances have affected ecosystem services provided by wetlands. Wetland assessments can be categorized into three levels based on how intensive and in-depth they are. Levels include remote sensing (level 1), rapid assessment (level 2), and in-depth (level 3); each of which has a variety of pros and cons.

1.4.1. Level 1 Assessments

Level 1 assessments are most commonly performed using remote sensing (Mita et al. 2007; Rooney et al. 2012). This allows the site to be assessed from a computer using a GIS database rather than a site visit, which are more expensive and time intensive (Mita et al. 2007; Rooney et al. 2012). This, in turn, allows more flexibility and efficiency in larger scale landscape planning and management (Mita et al. 2007). Some level 1 assessments, such as the Landscape Wetland Condition Analysis Model (LWCAM), have been found to have strong correlations to more in-depth level 3 assessments, therefore confirming their applicability for wetland assessments when an onsite visit is not logical (Mita et al. 2007). The major disadvantages of level 1 assessments are they only give the assessor a glimpse of the landscape condition and they are heavily reliant on previous land-use data which can be inaccurate (Hargiss et al. 2017). The quality of land-use data and software needs to be accurate and updated for level 1 methods to be effective.

1.4.2. Level 2 Assessments

Level 2 assessments include rapid assessment methods (RAMs), such as the North Dakota Rapid Assessment Method (NDRAM), and have been created to quickly and efficiently assess wetland conditions by quantifying a multitude of different biotic and land-use factors (Fennessy et al. 2007; Stein et al. 2009; Wigand et al. 2011; Hargiss et al. 2017). Fennessy et al. (2007) stated a rapid assessment should contain four criteria: it can be used to measure condition, is truly rapid, includes a site visit, and can be verified. To properly assess the ecological condition of a wetland it should be able to provide a single integrative score in a range relating to wetland condition. For a wetland assessment to truly be rapid it should take half a day or less. Otherwise, it should be considered a more in-depth level 3 assessment. A site visit should be required to ensure consistent and repeatable field protocols are followed by those assessing the wetland. Possibly the most important of the four criteria is the verification of its validity. To verify the accuracy in assessing wetland RAMs should be tested for a relationship with more in-depth level 3 methods, such as the HGM or a verified Index of Biological Integrity (IBI). Many of the RAMs seen today meet most, if not all, of the criteria to be considered a RAM. Some methods such as the Oklahoma Rapid Assessment Method, Delaware Method, Florida Wetland Rapid Assessment Procedure, Massachusetts Coastal Zone Management Rapid Habitat Assessment Method, Montana Wetland Assessment Method, Ohio Rapid Assessment method, North Dakota Rapid Assessment Method, and Washington State Wetland Rating System have all met the criteria proposed (Fennessy et al. 2007; Hargiss et al. 2009; Sifneos et al. 2010; Gallaway et al. 2020).

RAMs have been shown to not only accurately assess overall wetland condition, but also have strong correlations to aspects of certain biotic communities (Stein et al. 2009; Hargiss et al. 2017; Dupler et al. 2020; Gallaway et al. 2020). Dupler et al. (2020) found the Kentucky Wetland Rapid Assessment Method could be used to explain variation in richness of amphibian communities, as well as indicate the abundance of seven species of amphibians. Stein et al. (2009) and Gallaway et al. (2020) found significant relationship between RAMs and plant related data, including plant community composition, Floristic Quality Index scores, species richness, and diversity. This shows the applicability of RAMs goes beyond the general assessment of condition.

It has been shown training is of high importance for the consistency and repeatability of conducting RAMs and determining RAM scores (Herlihy et al. 2009). When properly trained assessors of all levels of expertise achieved more similar scores across multiple wetland types than experts with minimal training conducting RAMs across the same variety of wetland types. Training requires site visits, which means RAMs will be more expensive and time consuming than level 1 assessments, but still less so than more in-depth level 3 assessments (Stein et al. 2009; Wigand et al. 2011; Hargiss et al. 2017). Other downsides to RAMs are their subjective nature. They rely on best professional judgement of the assessor. However, with proper training, judgement decisions appear to be consistent across a range of expertise, and therefore, RAMs can be useful for sites requiring a visit where a more in-depth evaluation of wetland components is not necessary (Herlihy et al. 2009; Hargiss et al. 2017).

1.4.3. Level 3 Assessments

1.4.3.1. Hydrogeomorphic approach

One commonly used form of level 3 assessments is the Hydrogeomorphic approach (Hargiss et al. 2017). The Hydrogeomorphic approach (HGM) was developed by the United States Army Corps of Engineers and the National Resource Conservation Service. It uses a combination of soils data, landscape characteristics, and vegetation data to compare a particular wetland to reference condition sites (e.g. Gilbert et al. 2006 for the PPR). This method allows the surveyor to collect a large amount of data on multiple aspects of the wetland, which can be very useful for assessing wetland conditions (Hargiss et al. 2017). Since this method is used for mitigation purposes, it tends to be the most time-consuming method we commonly use in the PPR because of the exact nature and extent of data collected. It is therefore not always necessary when less time consuming and cheaper assessment methods can attain the same results.

1.4.3.2. IBIs

Studies on the IBI to assess wetland condition have looked at a variety of biotic subjects to quantify anthropogenic disturbance and overall wetland condition. The concept of an IBI was first proposed by Karr (1981) to assess riverine systems using fish populations. The term biological integrity can be defined as ‘‘the capability of supporting and maintaining a balanced integrated, adaptive community of organisms having a species composition, diversity and functional organization comparable to that of natural habitat of the region” (Karr and Dudley 1981). Subjects used to quantify wetland disturbance and conditions include vegetation (DeKeyser et al. 2003; Mack 2007; Hargiss et al. 2008; Rothrock et al. 2008), aquatic invertebrates (Tangen et al. 2003; Hanson et al. 2005; Anteau et al. 2011; Preston et al. 2018), birds (Jung et al. 2020), fish (Zimmer et al. 2000; Zimmer et al. 2002; Hanson et al. 2005; Herwig et al. 2010) and amphibians (Hossack et al. 2018; Smalling et al. 2019). Many IBIs, while very useful in certain contexts, have not undergone extensive testing to confirm their applicability (Mack 2007). However, when developed properly, IBIs can be efficient, cost-effective, and cover a fairly wide range of wetland types and ecosystems throughout a state or region (Fennessy et al. 2002; Mack 2007).

1.4.3.3. Plant Community IBIs

Floristic Quality Assessments (FQAs) or Floristic Quality Indices (FQIs) are some of the most commonly used condition assessment methods and are often incorporated into IBIs. They have been shown to be sufficient indicators of disturbance and overall wetland condition (Wilson et al. 2013; Gianopulos 2018). They provide in-depth information on the condition and function of a wetland, specifically as it relates to vascular flora present (TNGPFQAP 2001; Hargiss et al. 2017). FQAs generally are a measure of both the species richness and coefficients of conservatism (C-value) for those species within the specified region (Willhelm and Ladd 1988; Swink and Willhelm 1994; Taft et al. 1997; TNGPFQAP 2001). C-values are traditionally quantitative values (e.g. 0-10) assigned to each native species by a panel of experts on the region's flora based on the species' affinity toward natural areas and ability to handle disturbance and their presence along a disturbance gradient. Upsides of this method are its ability to provide valuable in-depth insight to the study site beyond what one would normally get from a level 1 or 2 assessment (Hargiss et al. 2017). Downsides include the training required to identify plant species correctly, the time-consuming nature of the survey, and the requirement of a site visit to conduct the assessment.

The use of FQAs frequently receives criticism. One criticism is based off the relation of the FQAs to a disturbance gradient reliant on the concept of reference condition wetlands for which there are differing definitions and lack in consistency on which definition to use within differing ecosystems (Lopez and Fennessy 2002; Stoddard et al. 2006; Otte et al. 2021). The concept of reference, and even general condition, becomes even more variable when thought of on a global scale rather than a regional scale (Otte et al. 2021). The phrase "reference condition" should be thought of in term of structure and function (Stoddard et al. 2006). The concept of reference condition should be narrowed down to a more specific definition. Stoddard et al. (2006) suggests the use of several definitions including minimally disturbed condition, historical condition, least disturbed condition, and best attainable condition. Due to the amount of disturbance and unlikeness of any historic condition wetlands remaining in the PPR, studies using FQAs and other condition assessments in the region should refer to reference conditions as least disturbed conditions or best attainable conditions. Once definitions are established, even determining reference measures for use of an FQA can be moderately subjective in regions where least disturbed or best attainable wetlands are rare or no longer present. Defining and categorizing reference wetlands changes across ecoregions is variable and can be unreliable across too wide of a study area (Herlihy et al. 2019). However, FQA scores needed to be at or near least disturbed or best attainable wetlands can be adjusted and variants can be added to fit the study area and type of wetlands present (Wilson et al. 2013; DeBerry and Perry 2015; Kutcher and Forrester 2018; Galloway et al. 2019). For the PPR, best attainable condition seems most fitting because no historical reference conditions remain but some native prairie areas remain fairly intact.

Another frequent criticism is their reliance on the subjectivity of C-values (Mushet et al. 2002; Matthews et al. 2015). Many species in certain groups such as perennial herbs, shrubs, and trees often get undervalued (Matthews et al. 2015). While some species C-values end up under or overvalued, as a whole the subjectivity of these values from experts does not seem to affect their ability to successfully play a part in assessing wetland condition through the use of FQAs (Mushet et al. 2002; Matthews et al. 2015). Panel-assigned C-values have, in some instances, even performed better than computer generated C-values; these computer values were generated using field data from both known reference and randomly selected sites (Mathews et al. 2015). In many cases, average C-values alone have been shown to be sufficient indicators of wetland condition, however, to ensure the accuracy and applicability of C-values they should be regularly updated when seen fit (Matthews et al. 2005; Bourdaghs et al. 2006; Bried et al. 2013).

Other criticisms of FQAs include that they lack the use of introduced species in many assessment calculations and their use of species richness, which is often influenced by wetland size and hydrogeomorphic class (Ervin et al. 2006; Kutcher and Forrester 2018). Including non-native species into FQA calculations could be considered due to the fact non-native species are often highly correlated to disturbance in a wetland. However, other studies have shown FQAs perform better without the incorporation of non-native species (DeBerry and Perry 2015). Species richness has been shown to increase with wetland size and longer water permanence (Kutcher and Forrester 2018). It has been proposed this could be offset by including non-native and native species abundance in addition to, or instead of, species richness. Overall, FQAs have been proven many times over to be correlated to, and therefore adequate for use in, wetland condition assessments (Wilson et al. 2013; Gianopulos 2018).

The Index of Plant Community Integrity (IPCI) was developed specifically for quantitatively assessing wetlands of the South Dakota, North Dakota, and Montana section of the PPR using their plant communities (DeKeyser et al. 2003). It was then refined and expanded by Hargiss et al. (2008) to be applicable over a larger area and a wider variety of hydrologic classifications. It takes into account 9 metrics related to species richness, community composition, and floristic quality (DeKeyser et al. 2003; Hargiss et al. 2008). Each individual metric is assigned a score (0, 4, 7, 11) based on value ranges for each metric. In addition to independently assessing wetland condition, the vegetative data collected from it can be incorporated into the more intensive HGM assessment to further certify results (DeKeyser et al. 2003).

There has been some criticism of the IPCI due to how it can be affected by inter-annual climate fluctuations (Euliss and Mushet 2011). They found condition rating to vary annually primarily due to changes in climate. This was likely caused by changes in hydrology due to annual variation in precipitation and evapotranspiration. This causes natural changes in species guilds, such as annual species which increased on mudflats in dry years. These mudflat annuals directly affect IPCI scores through the percentage of annual, biennial, and introduced species metric and indirectly through metrics related to floristic quality due to them being assigned lower C-values. They conclude by saying it is not always reproducible and has a high variability. However, this study had a much smaller scope than DeKeyser et al. (2003) and Hargiss et al. (2008) which developed the IPCI. Euliss and Mushet (2011) only assessed 16 wetlands with all 16 wetlands being in a very condensed and small area. The variability would have likely been much lower given a larger and more diverse sample size which could have experienced differing interannual climates.

The IPCI also takes more than wetland plant species into account by including the low prairie area (i.e. buffer area) in the survey effort and calculations which reduces the amount of variability coming from the plant communities in the center of the wetland being more heavily affected by climate (DeKeyser et al. 2003; Hargiss et al. 2008; DeKeyser et al. 2009). The low prairie area is less influenced by climate variations but can receive more influence from other disturbances such as fire, grazing, and proximity to agricultural land (DeKeyser et al. 2009). Regardless of variation within the metrics more heavily affected by climate, there are still six of the nine metrics which are not as highly correlated to shifts in climate (Euliss and Mushet 2011). The IPCI has and will continue to be a successful form of wetland condition assessment, and will have applicability for land managers in wetland monitoring, mitigation needs, restorations, and further research (DeKeyser et al. 2003; Hargiss et al. 2008).

More efficient methods of plant based bioassessments have been tested in recent years (Gianopulos 2018; Standen et al. 2018). Gianopulos (2018) showed FQAs in North Carolina could be simplified by removal of graminoids, which are the most difficult to identify, or using a dominance-based index where only the dominant species present are taken into consideration. This study showed both simplified measures appeared to be effective and correlated to RAM scores and full FQA values, while being much less time consuming and expensive. Even individual plant species have proven to be sufficient bioindicators of certain wetland conditions (Standen et al. 2018). *Sagittaria cuneata* (Arrowhead) is a an easily identifiable prairie wetland species which has been identified as an indicator of sediment enriched by nitrogen and phosphorus from agriculture due to unique changes in biomass and leaf shape when those nutrients are present. However, this method can only be used when *Sagittaria cuneata* is present. Many other efficient plant based bioassessments likely have still not yet been discovered but with increasing use, current and future methods will likely become even better and more efficient.

1.4.4. Multi-tier Methods

Multi-level methods have been a growing trend in recent years for wetland condition assessments (Stein et al. 2009; Hargiss et al. 2017). Whether used as validation for other methods or general wetland monitoring and assessment it has proven to be a useful tool for application throughout the PPR. The concept of a multi-tiered wetland assessment involves using multiple different levels of assessments ranging from a landscape assessment (level 1) to an intense assessment (level 3). Hargiss et al. (2017) tested the 3 levels of assessments for similarities in condition rankings to see which would be best used together to assess wetland conditions in North Dakota. The methods tested included the GIS based LWCAM (level 1), the NDRAM (level 2), the FQI (level 3), and the HGM (level 3). The LWCAM had the least similar condition ranking compared to other methods. The NDRAM, FQI, and HGM all gave very similar condition rankings to one another, showing a combination of any of the 3 could be used together. Each method differs drastically in time and resources needed to complete it. The LWCAM can be done remotely, the NDRAM requires a field visit but can be done in roughly 20 minutes, the

FQI can range from an hour to nearly a full day depending on the wetland, and the HGM model is the most time consuming of all the methods (DeKeyser et al. 2003; Gilbert et al. 2006; Hargiss et al. 2008; Hargiss et al. 2009; Hargiss et al. 2017). The FQI and HGM had the most similar condition/function rankings, but given the time allotments for conducting these assessments and the similarities in the scoring it makes the most sense to use the combination of the NDRAM and the FQI unless the more in-depth information of the HGM is needed for research or managerial (e.g. mitigation) purposes (Hargiss et al. 2017).

Generally, this use of multiple assessments involves different levels, but using multiple methods of the same level such as two IBIs (level 3) have proved useful as well (Wilson and Bayley 2012). In prairie wetlands of Canada, IBIs for wet meadow zone vegetation and wetland-dependent bird communities were shown to be strongly related and better indicators of environmental stress than any single biotic community alone. However, the use of two level 3 assessments arguably is not worth the extra time, cost, training, and other resources to justify their use when both individually can sufficiently be used to assess wetland condition along an environmental stress gradient. While use of more than two assessments could be done together, from a logistical standpoint it makes more sense to use the multi-tiered framework (Stein et al. 2009; Wilson and Bayley 2012; Hargiss et al. 2017).

1.5. Temporarily and Seasonally Ponded Wetland Plant Community Composition

Plant communities of PPR wetlands can be separated into distinct vegetation zones which are often correlated with water permanence, soil permeability, and groundwater within the potholes (Stewart and Kantrud 1971). The vegetation zones for temporarily and seasonally ponded potholes include an outer low prairie zone and an inner wet meadow zone, with a central shallow marsh zone for seasonally ponded wetlands. In many studies the wet-meadow zone is also often referred to as sedge meadow, especially regarding natural potholes (Mullhouse and Galatowitsch 2003; Myla et al. 2008). Different wetland types, such as fens, or more permanently ponded potholes may also include different or additional vegetation zones (Stewart and Kantrud 1971). Potholes experience distinct phases throughout the year, changing with changes in land use and water level fluctuations. These phases are known as normal emergent, open-water, drawdown bare-soil, natural drawdown emergent, cropland drawdown, and cropland tillage. Plant community composition often differs depending on the vegetation zone, phase, and salinity of the wetland. The distinction between these communities can range from being a clear and obvious boundary to a gradient to the adjacent community.

The low prairie zone consists of a narrow border around potholes serving as a buffer between the wetland and upland which is rarely inundated by water for periods of more than 1 to 2 days, other than right after snowmelt (Stewart and Kantrud 1971; Stewart and Kantrud 1972). It is comprised mostly of a combination of perennial upland grass, shrubs, and forb species such as *Poa pratensis* (Kentucky bluegrass), *Symphiocarpos occidentalis* (western snowberry), and *Ambrosia psilostachya* (cuman ragweed). The wet meadow zone is often inundated by water until well after snowmelt and regularly for several days after rainfall events, but never experiences high water levels. It is characterized by a mixture of perennial fine-textured grasses, rushes, sedges, and associated forbs such *Spartina pectinata* (prairie cordgrass), *Carex pellita* (wooly sedge), *Juncus arcticus* (mountain rush), and *Symphiotrichum lanceolatum* (white panicle aster). The shallow marsh zone maintains surface water through the spring, early summer, and sometimes even into the fall, but generally does not exceed 1 meter in depth. In the normal emergent phase species typically dominating the shallow marsh consist of medium to tall height grasses, grass-like, and forb species such as *Phalaris arundinacea* (reed canarygrass), *Carex* *atherodes* (wheat sedge), and *Sium suave* (hemlock waterparsnip). In the open water phase, it consists of submersed or floating aquatic forbs such as *Utricularia macrorhiza* (common bladderwort). The wet meadow and shallow marsh zones can also experience higher salinity conditions where dominant species will include those which have a higher tolerance to brackish conditions, such as *Hordeum jubatum* (foxtail barley). Drawdown phases are also common where inner zones can become dominated by different plant communities including mudflat annuals, such as *Chenopodium rubrum* (red goosefoot).

1.5.1. Natural Versus Restored Wetlands

Plant communities differ greatly between natural and restored prairie wetlands (Galatowitsch and van der Valk 1996a; Galatowitsch and van der Valk 1996b; Seabloom and van der Valk 2003a; Seabloom and van der Valk 2003b). Galatowitsch and van der Valk (1996a) determined restored wetlands tend to have significantly fewer species overall than their natural counterparts. Types of assemblages also differed greatly between the two. Natural wetlands had more wet-prairie, sedge meadow, shallow emergent, and floating annual species, while restored wetlands had greater number of deep emergent perennials, mudflat annuals, and woody plants. Natural wetlands also had higher species richness and seed density in the seed bank. Of the many species missing from restored wetlands, such as most sedges, most were also missing from the seed bank. This means those species are unlikely to reestablish themselves naturally in restored wetlands, with the exception of above ground dispersal, which is unlikely due to the isolated nature of many restored wetlands. While some species may naturally reestablish themselves within the early years of a restoration, they are often outcompeted by species such as *Phalaris arundinacea* (reed canarygrass) which establishes early and grows very quickly in restored wetlands.

Similarly, Seabloom and van der Valk (2003b) found differences in plant community composition overall, along elevational gradients, and in dominance of exotic species. They found more species to be distinctly associated with just natural wetlands than with restored wetlands. Species associated almost exclusively with natural wetlands were mostly native perennials, while species classified as restored wetland species were a combination of mudflat annuals, exotics, and perennials highly tolerant to disturbance. Natural wetlands appeared to have more evenly distributed species than restored wetlands. Even though both types had similar numbers of exotic species present, the distribution and dominance of those exotic species was much greater in restored wetlands. Overall vegetation cover, regardless of plant community composition, was much lower in restored wetlands. While there are many differences in plant community composition between natural and restored wetlands, research has shown the most evident and concerning differences appear to be the lack of a wet meadow and associated species along with the dominance of exotic species in restored wetlands (Galatowitsch and van der Valk 1996b; Seabloom and van der Valk 2003a; Seabloom and van der Valk 2003b).

1.5.2. Highly Disturbed Wetlands

A range of wetland disturbances including stormwater drainage, cultivation, ditching, and other land-use disturbance (e.g. urbanization) all have strong negative impacts on wetland plant communities (Galatowitsch et al. 2000). Overall species richness was not significantly different between disturbance types, but plant communities and guilds differed greatly depending on the type of disturbance. Alterations of hydrology due to stormwater runoff were shown to decreases native perennial plant cover while increasing cover of exotic perennials and floating vegetation. The dominant species present in stormwater wetlands were those tolerable to high levels of water fluctuations, such as *Phalaris arundinacea*. They also rarely contained species most characteristic of unimpacted sites. Similar to stormwater runoff, recent cultivation decreased the cover of native perennials, but also increased the prevalence and cover of annual species more so than exotic species. The effect of ditching was more dependent on surrounding adjacent land-use than anything else, with more highly disturbed land-uses, such as urbanization or agriculture, appearing to have a greater impact on species composition than ditching itself. Land-use disturbances impacted wetland plant communities most severely when within the immediate vicinity of the wetland and decreased with increasing distance from the site. The response and extent to which wetland plant communities are negatively impacted by anthropogenic disturbance varies, but all still experience guild shifts to less desirable plant communities in the wet meadow zone regardless.

1.6. Invasive Species

When it comes to invasive species for wetlands the term 'invasive' usually refers to non-native species with increasing abundance that displace native species after being introduced or in response environmental change (Bansal et al. 2019). The definition is not always clear as invasive species are context dependent and can be native, non-native, or hybrid species. Invasive species have many mechanisms by which they displace native flora. Some of those mechanisms include native patch suppression (Dillemuth et al. 2009), seedling resource suppression (Hager 2004; Vaccaro et al. 2009; Larkin et al. 2012), and soil nutrient and microbe modification (Jordan et al. 2008). Invasive species also often have a much higher tolerance to disturbance than native species, therefore becoming even more dominant in highly disturbed environments (Larson et al. 2001; Green and Galatowitsch 2002; Marlor et al. 2014; Grant et al. 2020b). Many invasive upland species such as *Bromus inermis* (smooth brome), *Poa pratensis* (Kentucky bluegrass), *Euphorbia esula* (leafy spurge), and *Cirsium arvense* (Canada thistle) can invade wetlands and displace native wetland species to an extent, but the most problematic invasive wetland species in the PPR appear to be *Phalaris arundinacea* (reed canarygrass) and *Typha x glauca* (hybrid cattail).

1.6.1. *Phalaris arundinacea* (reed canarygrass)

Phalaris arundinacea (reed canarygrass) is a species in the family Poaceae and is of growing interest and concern for research and management entities throughout North America due to its invasive nature throughout temperate wet habitats such as wetlands, lake shores, riverbanks, and floodplains (Lavergne and Molofsky 2004). It has become relevant to research and management efforts throughout the entire Midwest, and more locally the PPR, because of its pervasiveness and dominance as an invading species in wetlands (Galatowitsch et al. 1999).

Phalaris arundinacea is a long-lived perennial grass with a C3 photosynthetic pathway (Kephart and Buxton 1993). It can grow up to two meters tall and is highly rhizomatous, which contributes to its aggressive spread and invasive dominance (Katterer and Andren 1999). *Phalaris arundinacea* is mostly cross pollinated and has a high annual yield of seeds (Ostrem 1988a; Ostrem 1988b) which are most successfully germinated in water-saturated soils (Coops and Vandervelde 1995).

The origin of *Phalaris arundinacea* and how it became so established in North America has been widely disputed (Galatowitsch et al. 1999). There are native populations in both Eurasia and North America. Most believe the most prevalent population in the United States today may be a hybridization between North American and European populations. Regardless of its origins, *Phalaris arundinacea* populations seen today have been known to grow under a wide range of environmental conditions and disturbances. It has a wide variety of uses which led to its widespread distribution throughout North America. Some of the reasons it has been introduced throughout North America include its use as a forage, its ability to restore degraded soils and waters, and its use in bioenergy (Lavergne and Molofsky 2004; Krol et al. 2019).

Throughout most of its native and invasive range, *Phalaris arundinacea* most commonly occurs in wetlands, wet grasslands, and riparian areas (Galatowitsch et al. 2000). When present it can have very high vegetation coverage (often up to nearly 100%) due to its ability to form very dense stands. Coincidentally, native plant diversity is often also affected. When present, *Phalaris arundinacea* has been shown to significantly reduce plant diversity (Green and Galatowitsch 2002; Perry et al. 2004; Adams and Galatowitsch 2005; Schooler et al. 2006; Spyreas et al. 2010). This could lead to an increased risk of local extinctions of certain species in the future. It has also been shown to have negative effects on the floristic quality and abundance of present plant species in wetlands (Spyreas et al. 2010).

Not only does it affect the vegetation of wetlands, but also many of the ecosystem services they provide (Green and Galatowitsch 2002; Spyreas et al. 2010). Some of the ecosystem services of wetlands in the upper Midwest possibly affected include nutrient removal, filtration of sediments and chemicals, and wildlife habitat. Its effect on wildlife habitat has been of particular concern to researchers, natural resource managers, and people who use wetlands for recreational opportunities (Kirsch et al. 2007; Spyreas et al. 2010).

Insects have been seen to have the highest negative correlation to *Phalaris arundinacea* dominated landscapes (Spyreas et al. 2010). In localized studies both species richness and abundance of insects have been shown to decrease in relation to increased *Phalaris arundinacea* cover. This could also have negative side effects on other plant species in the area reliant on specific insects for pollination. The abundance and species richness of small mammals does not appear to have any relationship with *Phalaris arundinacea*. However, the abundance of specific species, such as voles and mice, are sometimes affected. Some bird species have been shown to be affected by *Phalaris arundinacea* dominance and cover (Kirsh et al. 2007). Certain species of breeding birds are shown to have a positive relationship, while others show a negative relationship, or no relationship, in regard to their use of wet meadows when *Phalaris arundinacea* is present (Kirsch et al. 2007; Spyreas et al. 2010). Some species may have a positive relationship or no relationship with abundance of *Phalaris arundinacea* because it is native to North America and often replaces other tall and lush plants with similar effects on bird species. Other bird species, large mammals, reptiles, and amphibians also are likely affected either directly or indirectly by *Phalaris arundinacea* dominance. Overall studies show biodiversity and biological integrity are being negatively affected by its dominance.

Mechanisms of *Phalaris arundinacea* invasion in wetlands are not completely understood but are often thought to include wetland nutrient enrichment, altered hydrology, altered soil chemistry, rapid growth, self-facilitation, and suppression of native seedlings (Galatowitsch et al. 1999). In wet meadows and shallow marsh areas, *Phalaris arundinacea* replaces native vegetation across wetlands of varying condition but is most common among those with high past or present disturbance levels, such as those currently in cultivated areas or those in areas previously cultivated and more recently restored (Galatowitsch et al. 2000). It preempts native vegetation establishment in restored wetlands of the upper Midwest. This is largely due to its rhizomatous and fast-growing nature (Adams and Galatowitsch 2005). It has a much faster growth rate in the first two years following germination which allows it to take up space and resources before other plant species. Its ability to take up above and below ground space before proper establishment of other species further facilitates its dominance and unconstrained growth.

Phalaris arundinacea establishment has been shown to be positively correlated to nitrogen availability in wetlands (Green and Galatowitsch 2002; Perry et al. 2004). Many wetlands in the northern Great Plains are in close proximity to agricultural land. These wetlands often have higher nitrogen levels due to agricultural runoff. In wetlands with excessive nitrogen levels *Phalaris arundinacea* has been shown to outcompete native sedges and grasses which would normally dominate wet meadow areas. Inversely, when nitrogen levels are low, sedge species are able to outcompete *Phalaris arundinacea*.

Another common mechanism of dominance is through its ability to take up aboveground space and limit light availability by forming dense stands and copious levels of litter accumulation (Galatowitsch et al. 2000; Schooler et al. 2006). These dense stands not only grow more quickly the first few years after germination, but often become established earlier in the year before other species can establish themselves. This limits the access of important resources, such as light, to native seedlings. Litter accumulation acts similarly to block light and use up space necessary for the growth of native seedlings. While native seedlings often cannot grow due to litter accumulation, other *Phalaris arundinacea* individuals are not as greatly affected, which allows litter accumulation to act as a mechanism of self-facilitation (Galatowitsch et al. 2000; Schooler et al. 2006; Vaccaro et al. 2009). These mechanisms of invasion are most prevalent when coupled with high levels of past or present disturbance (Galatowitsch et al. 2000).

Due to its aggressive nature in ecosystems with past and present disturbance, *Phalaris arundinacea* has posed a serious challenge to wetland managers in recent decades (Adams and Galatowitsch 2005; Adams and Galatowitsch 2006). In the Upper Midwest of the United States wetland wet meadows have been reduced to less than one percent of their former extent, so it is of crucial importance for wetland managers to maintain the biological integrity of the remaining wet meadow areas and reduce threats such as *Phalaris arundinacea* (Reuter 1986). *Phalaris arundinacea* management is particularly challenging to control in wetland restorations (Adams and Galatowitsch 2006). Because *Phalaris arundinacea* is often most dominant in disturbed areas, management efforts have mostly been focused on time frames around wetland restorations. Wetland restorations often focus on reestablishing wetland hydrology, but when *Phalaris arundinacea* is present the reestablishment of wetland hydrology does not diminish its persistence (Galatowitsch et al. 1999). Therefore, it is necessary to attempt to control *Phalaris arundinacea* before, during, and after restoration. It is most commonly controlled with spring burning and glyphosate application, which has not always been successful (Adams and Galatowitsch 2006). Spring burning was shown to be most effective. Burning is the spring was not shown to decrease the amount of *Phalaris arundinacea* present, but instead reduced its density in the seed bank, potentially limiting future recolonization. While traditionally herbicide application has been in the spring, after prescribed burns, in attempt to get the highest coverage of live shoots after litter is gone, fall herbicide treatments have been shown to be twice as effective.

Studies have shown burning in the spring and herbicide application in the fall is the most efficient method, but continuous treatments of wetlands using these techniques are still required to successfully control *Phalaris arundinacea* invasions in wetlands (Adams and Galatowitsch 2006). Managers should adequately prepare wetland sites over multiple growing seasons before restoration efforts and continue to monitor and selectively remove *Phalaris arundinacea* on sites after restoration. Efforts should also be put into place to monitor native grassland areas to ensure there is no encroachment of *Phalaris arundinacea* into wetlands in those areas as well. In wetlands with too much past or present disturbance and high cover of *Phalaris arundinacea*, when there is little to no chance of restoring the wetland to best attainable condition, other options such as use of *Phalaris arundinacea* for forage or biofuel should be considered.

1.6.2. *Typha x glauca* (hybrid cattail)

Typha x glauca (hybrid cattail) is a hybrid between species native and introduced to North America (Bansal et al. 2019). It is considered an invasive species throughout North American wetlands. Cattails were not prominent species in the PPR before the middle of the 20th century (Galatowitsch et al. 1999), with the earliest collection of *Typha x glauca* dating back to 1963 (DeKeyser, NDSU Herbarium Curator, personal observation). *Typha x glauca* can be distinguished from other species by the width of its leaves and the gap length between the upper male and lower female regions of its inflorescence. The large perennial forb is known primarily for its colonization of wetlands where it grows aggressively and rapidly into large monocultural stands. While *Typha aungustifolia* is also considered an invasive plant, genetic analyses of *Typha* in the PPR showed a majority of what remains in the region is the hybrid (Geddes et al. 2021). Therefore, for the purposes of this study, most *Typha* observations were recorded as *Typha x glauca* unless the observer was certain it was one of the other species.

In North Dakota cattail species were shown to be in as many as 49 percent of wetlands, where they covered 37 percent of wetlands when present, with those percentages likely increasing with increased anthropogenic disturbance drivers (Ralston et al. 2007). Its distribution and abundance are currently increasing in North America leading to management issues in attempting to control its spread and dominance (Bansal et al. 2019). It is able to spread great distances due to the wind dispersed nature of its seeds, and once established it can reproduce asexually through rhizomes. It has also increased due to anthropogenic disturbances such as alterations of wetlands hydrology, tillage, and increased nutrient loads from agricultural runoff.

Typha x glauca is able to thrive under anthropogenic disturbance whereas native species used to dominate under the natural disturbances of the region such as grazing and fire. Decreases in water levels and soil organic matter content and increases in soil bulk density of the PPR are consequences of these anthropogenic disturbances, which all favor *Typha x glauca* over native species.

The dense monocultures formed by *Typha x glauca* have a range of mostly negative but some positive impacts on wetland ecosystems (Bansal et al. 2019). It can negatively impact local flora, fauna, biogeochemical cycles, hydrology, and other functions. The dominance of *Typha x glauca* can both directly and indirectly impact the flora of wetlands. It has been shown to decrease diversity and abundance of native flora by outcompeting the normally native species, such as sedges (Larkin et al. 2012). Similar to *Phalaris arundinacea*, the size and density of *Typha x glauca* and its copious litter production have proven detrimental to native flora while minimally affecting other individual cattails, leading to a form of self-facilitation. *Typha x glauca* is able to occupy more permanently ponded regions of PPR wetlands which are normally occupied by submerged aquatic vegetation or open water (Greer et al. 2007; Bansal et al. 2019). *Typha x glauca* and its seeds are not edible to most fauna that frequent wetlands, and they displace species which waterfowl, fishes, and invertebrates normally use as a food source. Dense stands of *Typha x glauca* can also reduce habitat for amphibians. Another consequence of monotypic *Typha x glauca* stands is the effect on pollinators which stems from the reduction of native plant diversity needed for many specialized pollinators (Larson et al. 2006). *Typha x glauca* invasion tends to amplify nutrient cycling, specifically nitrogen and phosphorus cycles (Larkin et al. 2012; Bansal et al. 2019). This is largely due to it having greater nutrient uptake than native wetland plants. It also alters soil-nutrient biogeochemistry, due to its high litter production, as well as soil denitrification and methane emissions.

Despite the many negative consequences of *Typha x glauca* invasion, it also has the potential to increase specific ecosystem services if managed correctly (Bansal et al. 2019). Cattails have a variety of traditional uses ranging from forage for livestock, medicinal applications, productions of goods, kindling, and even food (Morton 1975). Other ecosystem services it can provide includes bioremediation, specifically phytoremediation, and use as a biofuel (Bansal et al. 2019). Phytoremediation from *Typha x glauca* is largely due to its rapid growth rate, tolerance of contaminated environments, and ability to uptake certain elements into its high aboveground biomass, which allows it to act as a nitrogen and phosphorus sink (Vamell et al. 2010; Hegazy et al. 2011). It can also be useful for the phytoremediation of agricultural chemical runoff and for metals accumulated within wetlands. It has also been suggested for use as a biofuel in wetlands where conventional agriculture is not likely to be successful because of reasons both beneficial for the wetlands and agriculture (Berry et al. 2017). It is quickly renewable due to its fast-growing nature, does not requiring normal agricultural processes such as tilling and replanting, and its removal is beneficial for wetland health. It shows promise as a bioenergy feedstock and a potential biofuel replacement for petroleum.

While there are potential beneficial uses for *Typha x glauca* the cost of invasion still greatly outweighs those benefits. It continues to be a concern for wetland management to this day. There have been a wide range of tactics attempting to control it in wetlands including herbicide treatment, water-level manipulation, hand-cutting, mowing, crushing, scraping, explosives, fire, drought, and altering salinity with none proving to be very successful (Bansal et al. 2019). Some treatments have proven to be moderately successful in the short-term but without active continuous management and repeated treatments no techniques have proven successful in the long-term.

Success of many *Typha x glauca* management regimes is often dependent on timing of treatment, wetland hydrology, and use of multiple treatments together (Bansal et al. 2019). Both draining of a wetland and altering water depth to exceed the tolerance of *Typha x glauca* has proven successful at times, especially when coupled with other treatments, but can also have negative consequences on other aspects of wetland ecology (Ball 1990). Control of *Typha x glauca* using herbicides has been successful for short-term management but can also lead to aggressive reinvasion and spread of herbicide resistant populations (Zheng et al. 2017). Use of fire as a treatment is successful in removing high amounts of *Typha x glauca* litter, which can be helpful for reestablishment of native species, but does not reduce its ability to rapidly grow back after fire (Smith and Kadlec 1985). Physical removal of *Typha x glauca* and the underlying sediment has proven successful in restoring wetlands to more natural plant communities, but it is difficult, time consuming, and expensive work which sometimes involves multiple physical removals (Smith et al. 2016). The harvest of *Typha x glauca* and its litter for bioenergy is one method proven to be fairly successful (Bansal et al. 2019). It is not a successful method for complete removal of *Typha x glauca* in wetlands, but biomass harvest both above and below water has been shown to significantly increase native plant diversity throughout the wetland. This is likely due to the lack of litter buildup associated with *Typha x glauca* stands when they are harvested. Although, for this method to be successful there needs to be relatively continuous and frequent harvest of these stands. Grazing by various species could also be an effective treatment but would also require continuous and fairly frequent grazing accompanied with other treatments to truly control it. Lastly, reducing nutrients that favor *Typha x glauca* (nitrogen and phosphorus) could prove useful for limiting *Typha x glauca* dominance (Elgersma et al. 2017). However, large-scale application of this method would prove challenging, especially in specific wetland types such as cropland wetlands. Ultimately, management of *Typha x glauca* within wetlands would require the utilization of multiple methods with specific regard to the sequence of those methods, scale, environmental conditions, and management objectives (Bansal et al. 2019).

There are an abundance of research needs regarding the *Typha x glauca* invasion in North America in order to gain a better understand of its ecology, genetics, uses, and how it should properly be managed (Bansal et al. 2019). There is a need to better understand how its distribution will spread with further anthropogenic disturbance and climate change. Results for this would likely come in the form of distribution modeling based on changing anthropogenic and environmental variables. Regional patterns of invasion need to be assessed in regard to genetics, specifically with non-native and hybrid species. More research is necessary for the ecological impact of invasive *Typha x glauca* on wetland trophic levels and foods webs within each specific region. Lastly, there is a need for further research on how *Typha x glauca* affects wetland mineralization, denitrification, rhizospheric-microbial relationships, methane emissions, and overall wetland carbon balance. This research will increase the overall understanding of the *Typha x glauca* invasion, and will help improve management decisions on how to control it and restore wetlands back to their more natural vegetative state.

1.7. Study Objectives

The FWS has been managing lands in the PPR for decades. The goals of the FWS have evolved to focus on diversity and ecological integrity of the lands they manage, therefore, management regimes have needed to evolve as well (Dixon et al. 2019). The proper management of wetlands on FWS fee-title lands is crucial to helping the FWS achieve its goals. Wetland management is complex and variable even on small scales, let alone a region-wide scale. This study looked to provide a base of relevant information pertaining to FWS fee-title land wetlands in the PPR.

The specific objectives of this study include: 1) assess the conditions of wetlands on FWS fee-title lands on a region-wide scale, 2) determine if specific site characteristics (e.g. native versus reseeded grassland) have a significant influence on wetland conditions, 3) provide insight into the most common plant species, the dominant plant species, and the overall plant community composition of wetlands on FWS fee-title land in the PPR, and 4) determine the major drivers influencing plant community composition. In this study, there was expected to be significant differences in site characteristics relating to land use and geographic setting but not between wetland classifications (e.g. seasonal and temporary). Plant communities were expected to be influenced by a combination of species-specific drivers (e.g. relative cover of *Typha x glauca*) and environmental factors (e.g. salinity). Results of this study can be used for aiding in future research and management aimed at improving the diversity and integrity of FWS fee-title lands.

2. FWS FEE-TITLE LAND WETLAND CONDITIONS

2.1. Methods

2.1.1. Site Selection

Wetland sample sites were selected using spatial data layers imported into a GIS environment then delineated across state, ecoregion, and FWS fee-title land boundaries (Tangen et al. 2019). The FWS National Wetlands Inventory was used to identify and classify wetlands. Only wetlands entirely within the boundaries of FWS fee-title land of North Dakota, South Dakota, and Montana were deemed usable for this study. Site selection was further constrained to only include seasonally and temporarily ponded wetlands within these boundaries. This resulted in 125 temporarily ponded and 125 seasonally ponded wetland sites for the study. Temporary and seasonally ponded wetlands were chosen because of their abundance, susceptibility to degradation due to disturbance, and importance for breeding waterfowl habitat. A generalized random tessellation stratified sampling design was used to generate a randomly selected but spatially balanced distribution of sampled potholes stratified by hydrologic regime and sample year (Stevens and Olsen 2004; Stevens and Jensen 2007). Sites were selected using the “spsurvey” package in R (Kincaid and Olsen 2019). This method of selection was used because it is more efficient than random sampling for populations that are unevenly distributed across the landscape. Sites were selected on FWS fee-title lands regardless of state, WMD, physiographic subregion, or land-use history.

After random selection of the 250 potholes, each was visually inspected using aerial imagery by a team of experts from the FWS, U.S. Geological Survey, and North Dakota State University (Tangen et al. 2019). If selected sites were deemed as not a pothole, not seasonally or temporarily ponded, within or connected to other systems, had disrupted hydrology, or not completely within the FWS boundary after visual inspection they were removed from the sample population and replaced with potholes from an oversample population. Of the 250 potholes selected 100 seasonally and 100 temporarily ponded wetlands were chosen as the primary sample sites with 25 of each classification remaining as oversample sites to be used when primary sites are deemed not appropriate for sampling by field crews. Field crews were also able to deem sites unusable upon inspection. Field crews would remove the site from the survey if the site was not a wetland (i.e. actually upland), not accessible (e.g. some sites were surrounded by private land), or were more permanent hydrology than seasonal (e.g. semi-permanent or permanent wetlands). Photo examples where sites met the field criteria to be deemed unusable and were replaced with oversample sites can be seen in Figure 2.1.

Figure 2.1. Wetlands deemed unusable for the purposes of the study and replaced with oversample wetlands. Figure Note: The image on the left was classified as a temporary wetland but showed no signs of hydrology or hydrophytic vegetation. The image in the right was classified as a seasonal wetland but deemed semipermanent. Both were replaced with wetlands from the oversample population.

The results of site selection reflected the relative abundance of potholes in each state with 157 in North Dakota, 91 in South Dakota, and two in Montana (Figure 2.2). Selection also reflected abundance regarding land-use with 176 being within reseeded grasslands (i.e. they were under cultivation at some point in time) and 74 within native remnant prairie. The distribution across subregions was 83 in the Glaciated Plains, 122 in the Missouri Coteau, 36 in the Prairie Coteau, 8 in the Lake Agassiz Plain, and 1 in the Turtle Mountains.

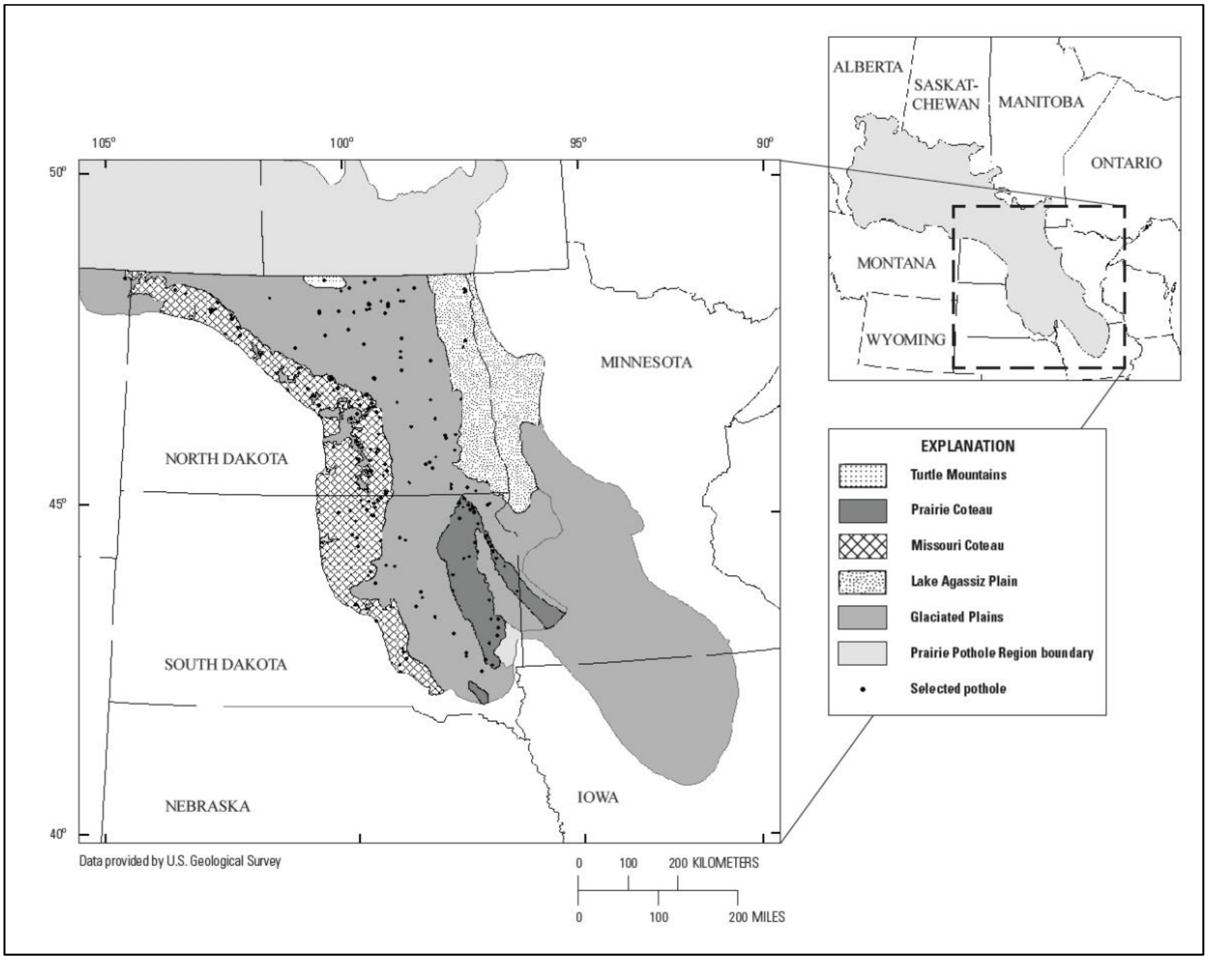

Figure 2.2. Prairie Pothole Region ecoregions used in this study and distribution of all 250 wetlands sites. Credit Tangen et al. (2019).

2.1.2. Assessment Methods

Sites were located using a combination of GPS location and a spatial data layer created in ArcGIS and transported to Google Earth, which can be used on mobile devices. Site assessments were completed during summer months (June-August) when most plants had flowered and were suitable for identification. Site assessments each year started with southernmost sites in South Dakota and ended with the northernmost sites of North Dakota and Montana. Each overall site assessment included an Index of IPCI survey, NDRAM survey, and the creation of a polygon of the estimated wetland vegetation boundary using ArcMap.

Upon arrival, each wetland was visually delineated to determine whether the wetland was temporarily or seasonally ponded, which have two and three primary vegetation zones respectively (Stewart and Kantrud 1971). Due to time constraints, number of vegetation zones present was the main method used to delineate wetlands. Wetlands originally classified as temporary but deemed seasonal, or vice versa, by field crews were assessed as they were classified by field crews, not by the original classification from the National Wetlands Inventory. Once wetland classification was determined field crews conducted an IPCI survey (DeKeyser et al. 2003; Hargiss et al. 2008). Primary species were determined for both temporarily and seasonally ponded wetlands using eight 1-square meter quadrats evenly distributed throughout the low-prairie zone and seven quadrats distributed throughout the wet meadow zone. Seasonally ponded wetlands also included five quadrats distributed throughout the shallow marsh zone. In each zone, quadrats were centered in the interior and exterior vegetation zones and oriented in a spiral pattern in the central zone (wet meadow in temporarily ponded and shallow marsh in the seasonally ponded wetlands). Layout for the quadrat method can be seen in Figure 2.3.

Within each quadrat, plant species were identified down to the species level (i.e. these were the primary species) when possible, and cover percentage of each species was estimated; along with cover of litter, standing dead plant material, and open water. Litter and water depth were also recorded for each quadrat. Secondary species were also identified and recorded between, but not within, each of the quadrats. Secondary species were not used for cover percentage like primary species but are still used in calculation of IPCI metrics. The time necessary to complete this survey depends on factors such as species diversity and size of the wetland, but generally ranged from 30 minutes to 5 hours per wetland.

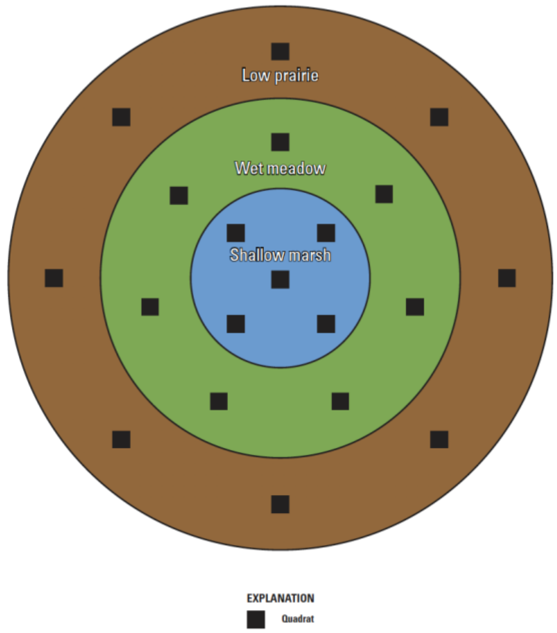

Figure 2.3. Quadrat layout method for the Index of Plant Community Integrity designed by DeKeyser et al. (2003) and Hargiss et al. (2008). Credit Tangen et al. (2019).

After completion of an IPCI assessment, field crews then completed an NDRAM survey. Completing the IPCI assessment first gave survey crews a better understanding of vegetation present at the site, which was useful when completing the NDRAM survey for wetlands (Hargiss 2009; Hargiss et al. 2017). The NDRAM determined wetland condition based on multiple metrics including buffers and surrounding land use (metric 1), hydrology and habitat alteration (metric 2), and vegetation (metric 3) (Appendix D). To complete this survey, field crews traveled around the wetland, completed a site description which includes information pertaining to vegetation, land-use, and hydrology, then scored multiple criteria for each metric based off that information. Criteria scored on the field data form includes buffer with and surrounding land intensity for metric 1; soil disturbance, habitat conditions, management, hydrologic effects, and potential to attain reference conditions for metric 2; and invasive species cover and overall condition of vegetation for metric 3. Each metric is scored numerically through categorization based on present and past stressors and trends toward recovery.

At each site field crews also created a polygon of the individual wetland boundary using a mobile version of ArcMap. The edge of the wet-meadow zone was determined by field crews and used as the wetland boundary. Once the boundary was established field crews used ArcMap, and walked the established boundary, while the program collected GPS data points following the route walked to create the polygon. A wetland boundary polygon was created at every site unless the wetland boundary could not be determined or there was a technical malfunction with ArcMap. When technical malfunctions with ArcMap boundary polygons were discovered, sites were revisited, and wetland boundaries were remapped if time restrictions allowed for it.

2.1.3. IPCI and NDRAM Scoring

Data collection by field crews followed the field survey methods outlined above for the IPCI and NDRAM. After data collection was completed, any necessary data was put into Microsoft Excel for organizational purposes. Scores for each metric of the NDRAM were summed to give the overall rapid assessment score and subsequently a wetland condition rating separating the wetland into one of four condition categories (0-26 for Poor; 27-52 for Fair Low; 53-68 for Fair High; and 69-100 for Good).

After data collection for the IPCI several steps needed to be taken to calculate the wetland condition score and rating. Wetland metrics required a complete species list for both the entirety of the wetland and the wet meadow zone by itself. Species lists were first separated by native or introduced species, then organized by life (perennial, biennial, annual), followed by alphabetical order. Every species list also includes the coefficient of conservatism (C-value) for each species provided by TNGPFQAP (2001). C-values range from 0-10 based off patterns of each plant species dependence upon natural areas to survive. Introduced species are not assigned a C-value. Once data was organized correctly, metric scores for each of the nine IPCI plant community attributes were calculated based on the metric value ranges determined by Hargiss et al. (2008) (Figure 2.4). Each of the nine plant community attributes were assigned one of four scores (0, 4, 7, and 11) to then be summed to give the overall IPCI score for the wetland (0-99). Due to differences in plant community composition temporarily and seasonally ponded wetlands are assigned different value ranges for each metric to receive each score given. The nine metrics assessed and the value ranges for each score can be seen in Figure 2.4.

Metric	0	4	7	11
Temporarily ponded				
Species richness of native perennials	0–16	17–23	24–40	≥41
Number of genera of native perennials	0–11	12–19	20–26	≥27
Number of native grass and grass-like species	0–8	9–10	11–15	≥16
Percentage of annual, biennial, and introduced species	≥41.1	35.1–41.0	27.1–35.0	0.0–27.0
Number of native perennial species in wet-meadow zone	0–7	8–10	11–13	≥14
Number of species with C value ≥5	0–4	5–11	12–16	≥17
Number of species in the wet-meadow zone with C value ≥4	0–3	4–9	10–12	≥13
Average C value	0.00–2.50	2.51–3.57	3.58–4.58	≥4.59
FQI	0.00–13.60	13.61–21.70	21.71–27.20	≥27.21
Seasonally ponded				
Species richness of native perennials	0–19	20–31	32–41	≥42
Number of genera of native perennials	0–14	15–24	25–32	≥33
Number of native grass and grass-like species	0–6	7–10	11–17	≥18
Percentage of annual, biennial, and introduced species	≥41.1	30.8–41.0	21.1–30.7	0.0–21.0
Number of native perennial species in wet-meadow zone	0–8	9–16	17–24	≥25
Number of species with C value ≥5	0–7	8–17	18–26	≥27
Number of species in the wet-meadow zone with C value ≥4	0–4	5–9	10–16	≥17
Average C value	0.00–2.60	2.61–3.12	3.13–3.52	≥3.53
FQI	0.00–10.00	10.01–16.11	16.12–22.99	≥23.00

Figure 2.4. Index of Plant Community Integrity metric value ranges and scores. Credit Hargiss et al. (2008).

Average C-value and FQI calculations both require basic mathematical calculations based on C-Values. Average C-value is calculated by summing the C-values for native species present in the survey and dividing that by the total number of native species present (TNGPFQAP 2001).

$$\bar{C} = \frac{\Sigma C}{N}$$

The FQI is a weighted species richness estimate IBI used in the IPCI to discriminate between sites that have similar average C-values but differ significantly in number of native species present. It is calculated by multiplying the average C-value of native species present by the square root of the number of native species present.

$$FQI = \bar{C}\sqrt{N}$$

Once C-value and FQI metrics were calculated, all metrics were summed to give the overall IPCI score (0-99) of the wetland (Hargiss et al. 2008). Scores were used to assign each wetland a condition rating. Condition rating scores differ between temporarily and seasonally ponded wetlands with temporarily ponded wetlands having 3 condition ratings (0-33 for poor, 34-66 for fair, and 67-99 for good) and seasonally ponded wetlands having 5 condition ratings (0-19 for very poor, 20-39 for poor, 40-59 for fair, 60-79 for good, and 80-99 for very good).

Once all IPCI and NDRAM scores were calculated all scores and corresponding wetland data was organized in Microsoft Excel. All IPCI and NDRAM scores were averaged, highs and lows recorded, and condition rating categories were summed to provide a general summary of all 200 wetlands. Each of the 9 IPCI metrics and total species diversity by site were also averaged for all 200 wetlands.

2.1.4. Data Analysis

Species richness, NDRAM scores, and IPCI scores were examined using the Generalized Linear Model (GLM) procedure in SAS Enterprise Guide 7.1 (Copyright © 2017 by SAS

Institute Inc. Cary, NC, USA). Three separate models were used to determine whether Species richness, NDRAM score, or IPCI score responded to wetland classification (seasonal versus temporary), grassland type (native versus reseeded), or an interaction term between the two (classification x grassland type). When ANOVA (Analysis of Variance) yielded a significant model, Tukey's Studentized Range (HSD) test was used to make pairwise comparisons of wetland classification, grassland type, and the interaction term. When the interaction term yielded a significant model Least Square Means was used to make multiple comparisons.

One-way analysis of variance for wetland WMDs used the ANOVA procedure to identify significant differences ($p \leq 0.05$) between the 16 WMDs. Three separate models were used to determine whether species richness, NDRAM scores, or IPCI scores responded to WMDs. The Tukey's Studentized Range (HSD) test was used to make comparisons for every WMD.

2.2. Results

In total, 200 wetlands were assessed. Of the wetlands assessed, 100 were classified as temporary and 100 were classified as seasonal by the National Wetlands Inventory (NWI). However, field crews had the ability to change the classification based on the number of wetland vegetation zones they observed or hydrology present. Classification changes can be seen in Table 2.1 below. Any wetlands which changed classification to upland, ephemeral, or semipermanent were replaced with oversample wetlands but any which remained temporary or seasonal after a classification change were still assessed. This resulted in 59 wetlands assessed as temporary and 141 assessed as seasonal. Methods for detecting and classifying wetlands have greatly improved since the original NWI classifications. The NWI, while still useful, is frequently incorrect in its classifications as seen here. Even though technological advancements have improved, a site visit remains the most dependable method to determine wetland classification. Assessed wetland sites and site characteristics can be seen in Appendix B.

Table 2.1. Wetland classification changes from field observations.

Change in Classification	Number of Wetlands
Temporary to Upland/Ephemeral	12
Temporary to Seasonal	52
Seasonal to Upland/Ephemeral	2
Seasonal to Temporary	11
Seasonal to Semipermanent	6

2.2.1. Species Richness

Species richness ranged from 12, for a temporary wetland on reseeded grassland in the Sand Lake WMD, to 96 for a seasonal wetland on native grassland in the Waubay WMD with the overall mean being 38.17. The mean species richness for the 48 native grasslands assessed was 49.58 and 34.56 for the 152 reseeded grasslands. Mean species richness for the 141 seasonal wetlands assessed was 42.17 and 28.57 for the 59 temporary wetlands. Crosby WMD had the highest mean species richness (53.33) while Sand Lake had the lowest (28.4).

The GLM procedure with the Tukey's Studentized Range (HSD) test showed significant difference in species richness for wetland classification ($p=0.0328$), grassland type ($p<0.0001$), and the interaction term ($p=0.0094$). There was a significant difference for all comparisons of the interaction term except for temporary wetlands on native grassland versus both temporary wetlands on reseeded grassland and seasonal wetlands on reseeded grassland. Least Square means for the interaction term of wetland classification and grassland type p-values can be seen in Table 2.2.

Table 2.2. P-values from Least Square Means of the GLM Procedure with the Tukey's Studentized Range (HSD) test to adjust for multiple comparisons of species richness.

	Seasonal Reseeded	Seasonal Native	Temporary Reseeded	Temporary Native
Seasonal Reseeded		<.0001	<.0001	0.5079
Seasonal Native	<.0001		<.0001	<.0001
Temporary Reseeded	<.0001	<.0001		0.2589
Temporary Native	0.5079	<.0001	0.2589	

One-way analysis of variance for WMDs using the ANOVA procedure with the Tukey's Studentized Range (HSD) test showed significant differences ($p \leq 0.05$) between WMDs for species richness. Significant ($p \leq 0.05$) differences were found for species richness between the Waubay WMD and the following WMDs: Kulm, Tewaukon, Devils Lake, Huron, and Sand Lake. Significant differences ($p \leq 0.05$) were also found between the Lostwood WMD and the Sand Lake WMD.

2.2.2. NDRAM Scores

NDRAM scores ranged from 34 for a seasonal wetland on reseeded grassland in the Valley City WMD to 98 for a seasonal wetland on native grassland in the Waubay WMD, with the mean being 57.33 for all 200 wetlands surveyed. The mean score for the 48 wetlands on native grassland was 72.75 and 52.46 for the 152 wetlands on reseeded grassland. The mean score for the 141 seasonal wetlands assessed was 57.83 and 56.14 for the 59 temporary wetlands. The Lostwood WMD had the highest mean score (74.46) while the Arrowwood WMD had the lowest (50.75). Of the 200 wetlands 0 were rated as poor, 66 as fair low, 102 as fair high, and 32 as good.

The GLM procedure with the Tukey's Studentized Range (HSD) test showed significant difference in NDRAM scores for wetland classification ($p<0.0001$), grassland type ($p<0.0001$), and the interaction term ($p=0.0035$). There was a significant difference for all comparisons of the interaction term except for seasonal wetlands on reseeded grassland versus temporary wetlands on reseeded grassland. Least Square means for the interaction term of wetland classification and grassland type p-values can be seen in Table 2.3.

Table 2.3. P-values from Least Square Means of the GLM Procedure with the Tukey's Studentized Range (HSD) test to adjust for multiple comparisons of NDRAM Scores.

	Seasonal Reseeded	Seasonal Native	Temporary Reseeded	Temporary Native
Seasonal Reseeded		<.0001	0.9597	<.0001
Seasonal Native	<.0001		<.0001	0.0359
Temporary Reseeded	0.9597	<.0001		<.0001
Temporary Native	<.0001	0.0359	<.0001	

One-way analysis of variance for WMDs using the ANOVA procedure with the Tukey's Studentized Range (HSD) test showed significant differences ($p \leq 0.05$) between WMDs for NDRAM scores. Significant ($p \leq 0.05$) differences were found for NDRAM scores between the Waubay WMD and the following WMDs: Kulm, Chase Lake, Sand Lake, Devils Lake, Tewaukon, J. Clark Slayer, and Lake Andes. Significant differences ($p \leq 0.05$) were also found between the Lostwood WMD and the following WMDs: Huron, Kulm, Chase Lake, Sand Lake, Valley City, Devils Lake, Tewaukon, J. Clark Slayer, Arrowwood, and Lake Andes.

2.2.3. IPCI Scores

IPCI scores ranged from 0 to 99 with the mean being 40.91. In total 11 different wetlands received a score of 0 while only 3 received the highest score of 99. The mean score for the 48 wetlands on native grassland was 65.13 and 33.26 for the 152 wetlands on reseeded grassland. The mean score for the 141 seasonal wetlands assessed was 46.46 and 27.63 for the 59 temporary wetlands. The Crosby WMD had the highest mean score (72.67) while the Lake Andes WMD had the lowest (23.6). Of the 200 wetlands 14 were rated as very poor, 85 as poor, 50 as fair, 36 as good, and 15 as very good. The averages for each of the nine IPCI metrics were also calculated and can be found in Table 2.4.

Table 2.4. IPCI averages for all 9 IPCI metrics across all wetlands assessed in 2020 and 2021.

Metrics	Average
Species richness of native perennials	25.74
Number of genera of native perennials	21.7
Number of native grass and grass-like species	8.5
Number of annual, biennial, and introduced species	12.43
Number of native perennials in wet meadow zone	14.86
Number of species with C-value ≥ 5	8.13
Number of species in wet meadow zone with C-value ≥ 4	8.29
Average C-value	3.35
FQI	17.72

The GLM procedure with the Tukey's Studentized Range (HSD) test showed significant difference in IPCI scores for wetland classification ($p<0.0001$) and grassland type ($p<0.0001$). There was not a significant difference in IPCI scores for the interaction term ($p=0.0661$). Since no significant difference was found for the interaction term it was not necessary to run the GLM procedure with Least Square Means.

One-way analysis of variance for WMDs using the ANOVA procedure with the Tukey's Studentized Range (HSD) test showed significant differences ($p\leq0.05$) between WMDs for IPCI scores. Significant differences ($p\leq0.05$) were found for IPCI scores between the Waubay WMD and the following WMDs: Devils Lake, Huron, and Sand Lake. Significant differences ($p\leq0.05$) for IPCI scores were also found between Lostwood and Sand Lake WMDs.

2.3. Discussion

2.3.1. Species Richness, IPCI Scores, and NDRAM Scores

Analysis of species richness and NDRAM score data showed significant differences ($p\leq0.05$) for at least some categories of all variables tested. Analysis of IPCI scores showed significant differences between grassland type (native versus reseeded), classification (seasonal versus temporary), and some WMDs, but not the interaction term (grassland type x classification). Significant differences were expected between grassland types and between some of the WMDs but were not expected between seasonal and temporary wetlands.

This study confirmed the results of previous studies showing natural or native wetlands have higher species richness than restored wetlands (Seabloom and van der Valk 2003b). The mean species richness was 49.58 for wetlands in native grasslands and only 34.56 for wetlands in reseeded grassland. The percentage of those species being annual, biennial, and introduced also tended to be higher for wetlands in reseeded grasslands. In this study, significantly more wetlands in reseeded grassland were assessed than wetlands in native grassland, but a majority of native plant species encountered were from assessments on native grassland. Most desired native species were rarely found in wetlands on reseeded grassland and were usually in low abundance when they were. In contrast, even though some invasive species such as *Phalaris arundinacea* (reed canarygrass) and *Cirsium arvense* (Canada thistle) were found during almost every wetland assessment, they were often sparse in native grassland areas and abundant in reseeded grassland. This likely influenced species richness differences between the two.

Reseeded grasslands are those which used to be predominantly used for agriculture but were reseeded after acquisition (Dixon et al. 2019). Wetlands on these lands were often cultivated and possibly drained for years and sometimes even decades. If wetlands were dry enough, they were reseeded with a plant mixture that was designed for the adjacent uplands. Historically, this may have included grass species that are considered invasive today (e.g. *Phalaris arundinacea* and *Bromus inermis*). Current upland seed mixtures are composed of native grasses, forbs, and shrubs. Few FWS fee-title land wetlands were reseeded with specific native wetland plant species. Instead, they relied heavily on natural recolonization. In reseeded wetlands, even though species richness often increases significantly after reseeding the species accumulation rate eventually levels out (Myla et al. 2008). This means these wetlands become unlikely to continue supporting additional species necessary to resemble wetlands in native grassland.

Many of the reseeded grasslands are isolated on the landscape and still predominantly surrounded by agriculture in many parts of the PPR. This means desired native species which were not part of the reseeding must travel far distances to reach many of the reseeded grasslands in order for natural recolonization to occur. Instead, it has been found that the results of natural recolonization vary and often do not appear successful for reestablishing native plant communities. Mullhouse and Galatowitsch (2003) found naturally colonized wet meadow areas were made up of primarily invasive perennial species, such as *Phalaris arundinacea* (reed canarygrass), which often approached 75-100% in cover. Many wet meadow species commonly found in wetlands of native grasslands were completely absent or found in only low abundances. This was also seen in Seabloom and van der Valk (2003b) where lower species richness and lack of desired native flora was found to be primarily caused by dispersal limitation for restored wetlands.

Many wetlands in reseeded grassland have previously disturbed soils from past cultivation of the land. While the innermost zones of many wetlands were often wet enough where they could not be cultivated, the same cannot be said for the wet meadow zones. Due to this, wet meadow zones were often cultivated as well (Seabloom and van der Valk 2003a). This resulted in less well developed or even completely lacking wet meadow zones in restored wetlands. This would also reduce species richness as many species are dependent on a well-developed wet meadow zone to survive.

Another consequence of former soil disturbance is that invasive species have adapted to tolerate higher levels of disturbance such as this (Galatowitsch et al. 1999). Invasive species can often still be found in cultivated or urban wetland areas where more desirable native species cannot (Galatowitsch et al. 2000). With many reseeded grasslands initially being seeded into invasive grass species, and being surrounded by agriculturally dominated landscapes, there is an easier path for those invasive species to naturally recolonize wetlands in reseeded grasslands than desirable native species, whose seeds may have to travel further distances (Seabloom and van der Valk 2003b). Even when the seeds from desirable native species can reach reseeded grasslands, they are often outcompeted by other species which are already dominating the wetlands. The high cover of these invasive species and their litter, along with the inability of desired native species to recolonize naturally, means many of the wetlands in reseeded grasslands will have a much lower total species richness compared to those in native grasslands. Even wetlands that have been properly restored, reseeded with native species, and monitored tend to have significantly fewer species than wetlands in native grassland (Galatowitsch and van der Valk 1996a).

Significant differences in species richness between seasonal and temporary wetlands were not expected (see Figure 2.4). This is likely a result of a combination of factors. Seasonal wetlands tend to be larger in size than temporary wetlands which generally results in increased species richness (Kutcher and Forrester 2018). Larger size means more overall area in and around the wetland for species to inhabit and an increased chance of encountering small patches of higher diversity even if the overall cover of the dominant species is not significantly different.

Another likely reason is that seasonal wetlands have an additional vegetation zone (shallow marsh zone) which is inhabited by a completely different set of species than the other zones (Steward and Kantrud 1971). The shallow marsh zone is often the least diverse of the 3 zones but can still increase the overall species richness by including species which cannot be found in temporary wetlands.

The reasons for significant differences between the interaction terms are likely the same as the previous reasons discussed regarding native versus reseeded grassland and seasonal versus temporary wetlands. However, as seen in Table 2.3, two of the interaction terms were not significantly different; temporary wetlands on native grassland did not have significantly different species richness than temporary wetlands on reseeded grasslands or seasonal wetlands on reseeded grassland. Average species richness was 33.38 for temporary wetlands on native grassland, 27.22 for temporary wetlands on reseeded grassland, and 37.75 for seasonal wetlands on reseeded grassland. Temporary wetlands on native grassland may have different plant communities but likely due to being smaller in size and not having a shallow marsh zone they have a similar species richness to seasonal wetlands on reseeded grassland.

Similarity in species richness between grassland types for temporary wetlands may be due to temporary wetlands being less affected by the surrounding landscape or past land use disturbance, but it is unlikely. More likely it indicates the methods were not sensitive enough to detect differences, wetland types are in fact similar, or temporary wetlands are more susceptible to changes in climate altering wetland functions and plant communities. Climate changes can greatly alter the hydrology of temporary wetlands which disrupts their normal seasonal, annual, and interannual vegetation cover cycles (Euliss et al. 2004; Johnson et al. 2016). These cycles are largely responsible for species richness and composition in prairie pothole wetlands. Climate changes could be disproportionately affecting wetland hydrology and vegetation cover cycles in the smaller less wet temporary wetlands. It is also possible temporary wetlands are more susceptible to invasive species which could outcompete many of the other species and reduce species richness. It seems most likely that a combination of these factors is affecting temporary wetlands in this study.

Tangen et al. (2019) provided a sampling design based on the location of FWS fee-title lands, which are not equally distributed across ecoregions (Figure 2.2). Analyses of FWS administrative areas, or WMDs, provided appropriate sample sizes to potentially yield useful insights. Differences in WMDs also has the potential to reflect differences in management regimes, not just geographic location. Significant differences found among WMDs, conceivably are due to past and present land uses, respective ecoregion, and possibly even historical and present-day management resources. The past and present land use and availability of management resources in the WMDs are largely dependent on location within North and South Dakota. The Waubay WMD had significantly higher species richness than the Kulm, Tewaukon, Devils Lake, Huron, and Sand Lake WMDs. Nearly all wetlands assessed in the Waubay WMD were within the Prairie Coteau ecoregion which is comprised of glacial sediment deposits forming a more hilly and rocky landscape (Bluemle 2000). Due to this much of the area was not cultivated in the past but rather used as rangeland for cattle to graze which is how it is still largely used today. Grazing with cattle has been shown to increase species richness (Larson et al. 2020). Less previous cultivation also means a higher percentage of wetlands assessed would be in native grassland areas. Of the 18 Waubay wetlands assessed, 12 were in native grassland as compared to the WMDs with significantly lower species richness: 1 of 22 for Kulm, 1 of 10 for Tewaukon, 4 of 31 for Devils Lake, 5 of 10 for Huron, and 1 of 20 for Sand Lake. These WMDs are largely within the southern end of Missouri Coteau and the Glaciated Plains ecoregions. The Missouri Coteau landscape is similar to the Prairie Coteau, but the Glaciated Plains is more gently sloping (Bluemle 2000). As seen by the numbers of wetlands assessed in native grassland, these WMDs were much more dominated by agricultural in the past. They also continue to be in an agriculturally dominated landscape today which limits native plant dispersal (Seabloom and van der Valk 2003b), increases dominance of invasive species (Galatowitsch et al. 1999), and decreases the access to cattle for grazing the fee-title lands which can greatly affect species richness (Larson et al. 2020).

Significant difference between the Lostwood WMD and the Sand Lake WMD were identified. Lostwood WMD is located at the northern end of the Missouri Coteau which has a hilly landscape with a large quantity of native grassland remaining (Bluemle 2000). Of the wetlands assessed in the Lostwood WMD 10 of 13 were in native grassland as compared to only 1 of 20 for the Sand Lake WMD. These differences in native grassland do not necessarily represent the actual ratio of native grassland to former cropland within these WMDs, rather, the sites assessed are a result of the Tangen et al. (2019) design. According to Smith (2020), targeted prescribed fire occurred on a regular basis from 1978-2001 at Lostwood WMD, whereas records do not indicate the same as Sand Lake.

The NDRAM scores were much less variable than IPCI scores and species richness. There were no wetlands that received a condition rating of poor according to NDRAM scores. This should be accurate seeing as the NDRAM was developed taking the most disturbed wetland sites, such as those drained or in agricultural fields, into consideration (Hargiss et al. 2009). The NDRAM takes many landscape factors into consideration; not just vegetation like the IPCI and species richness. There were still extreme differences in scores and condition ratings between the most severely degraded and most intact wetlands assessed showing the NDRAM is still a viable and accurate method for assessing wetland condition in the field without the need for plant identification expertise.

Similar to species richness, analysis of NDRAM scores showed significant differences ($p \leq 0.05$) for at least some categories of all variables tested: grassland type, wetland classification, the interaction term (grassland type x wetland classification), and WMDs. Significant differences were expected for at least some aspects of all comparisons except for wetland classification. Significant differences in grassland type (native versus reseeded) were expected because many of the land use and habitat related metrics and categories used to assess and score wetlands using the NDRAM are directly related to the past and present land use disturbance (e.g. tillage). Native grasslands have less past and present land use disturbance than reseeded grasslands which likely still have not fully recovered from past disturbances such as agriculture. Therefore, NDRAM scores for native grassland tend to be higher regardless of the metrics incorporated which are independent of the past and present land use.

Significant difference in wetland classification (seasonal versus temporary) was not expected for NDRAM scores. The differences seen are likely due to the metrics used for NDRAM scores that are independent of land use, such as the vegetation related metrics and categories. As seen in the analysis of species richness, temporary wetlands had a significantly lower species richness than seasonal wetlands. Seeing as NDRAM assessments were completed after IPCI assessments where the species richness and species guilds were observed, temporary wetlands would receive lower scores for categories relating to vegetation due to lower species richness and higher cover of invasive species.

The interaction term (grassland type x wetland classification) for NDRAM scores can be seen in Figure 2.3. There were significant differences between NDRAM scores for all comparisons except for temporary wetlands on reseeded grassland versus seasonal wetlands on reseeded grassland. This is likely due to both receiving similarly low scores for land use and habitat alteration metrics, with vegetation not being different enough to counteract the similarity. While seasonal wetlands in reseeded grasslands had significantly higher species richness, the cover of invasive species likely remained similar enough to where vegetation metrics were not scored significantly different. Significant differences in the rest of the comparisons are likely the result of a combination of the reasons leading to significant differences between grassland types and wetland classifications independently.

One way analysis of variance for WMDs showed significant differences ($p \leq 0.05$) were found for NDRAM scores between the Waubay WMD and the following WMDs: Kulm, Chase Lake, Sand Lake, Devils Lake, Tewaukon, J. Clark Slayer, and Lake Andes. Significant differences were also found between the Lostwood WMD and the following WMDs: Huron, Kulm, Chase Lake, Sand Lake, Valley City, Devils Lake, Tewaukon, J. Clark Slayer, Arrowwood, and Lake Andes. These differences are likely due to various factors, including the presence of more cropland in certain WMDs, climate variation, and the rations of native grasslands to former croplands surveyed. For example, Waubay and Lostwood WMDs are largely located in areas where there are still large sections of native grassland due to the larger hills and rockier landscape in those areas (Bluemle 2000). This means less cultivation on adjacent croplands and more access to cattle for grazing as well as prescribed fire. These factors would also have had a positive influence on the vegetation of the wetlands in these areas which are also used as for metrics in scoring the NDRAM (Galatowitsch and van der Valk 1996a; Hargiss et al. 2009; Larson et al. 2020)

The IPCI scores had a wider range and higher standard deviation than species richness and NDRAM scores. There were 99 wetlands to receive the IPCI rating of poor or very poor and 11 to receive the lowest possible IPCI score of 0, while 51 were rated as good or very good and 3 received the highest possible score of 99. Wetlands receiving high scores and good ratings were almost always in native grassland areas with little disturbance (e.g. tillage), but those receiving low scores and poor ratings were not always necessarily in the areas of highest disturbance, such as agricultural fields. Given the quantity of low scores in areas which would not appear at the lowest end of a disturbance gradient, IPCI scores appeared not to always follow the disturbance gradient and condition continuum put forth in the development of the IPCI scores, but rather follow the condition ratings more closely (DeKeyser et al. 2003; Hargiss et al. 2008). This could either mean the metric value ranges for IPCI scores need to be recalibrated to reflect the current disturbance gradient across all ecoregions in the study area or that many wetlands, especially those on reseeded grasslands, are unable to support a diverse mixture of plant species typical of PPR wetlands. It is likely that the latter is true, and being dominated by a few invasive species has deterred the development of a resilient, diverse plant community in the reseeded grassland areas. Disturbance and high cover of invasive species has been linked to less desirable plant communities (Galatowitsch et al. 2000; Mullhouse and Galatowitsch 2003) which would result in lower IPCI scores.

Similar to species richness and NDRAM scores, there were significant differences between IPCI scores for grassland type, wetland classification, and WMDs. However, there was not a significant difference for the interaction term (grassland type x wetland classification). This likely means when more factors were taken into consideration IPCI scores became more similar, to the extent they were no longer significantly different. Significant differences between IPCI scores for grassland type, wetland classification, and WMDs are likely due to the same reasons for significant differences in species richness. Of the nine metrics used to calculate IPCI scores, six involve some aspect of species richness while the other three are centered around C-values (DeKeyser et al. 2003; Hargiss et al. 2008). High species richness generally leads to a higher IPCI score while low species richness leads to a lower score. One exception is when a high percentage of the species present are annuals, biennials, or introduced species. For future research it may be redundant to analyze both IPCI scores and species richness. IPCI metric scores or values could also be analyzed individually to determine which metrics are and are not significantly different between grassland type, wetland classification, and WMDs.

2.3.2. Conclusion

This was a baseline study to determine the conditions of wetlands on FWS fee-title land across the PPR and what factors influenced those conditions most. The main factors appearing to influence PPR wetland species richness, NDRAM scores, and IPCI scores appears to be past and present land use, the presence of adjacent or nearby cropland, and consequently native species seed dispersal limitation, access to land management resources (i.e. burning and grazing), invasive species dominance, and possibly wetland size. Analyzing species richness and IPCI scores would seem redundant but different results for the interaction term of grassland type and wetland classification shows analyzing both can still be a useful method.

Overall, the difference between seasonal and temporary wetlands is significant, but wetlands in native grassland versus reseeded grassland seems to be the most distinct. Even many years after restoration, reseeding, and continued management these reseeded wetlands are still in much worse condition than their native counterparts. The fact they still do not resemble native wetlands for species richness, NDRAM scores, or IPCI scores means most of them likely never will. However, they still remain crucial and necessary parts of the ecosystem because of the many ecosystem services they provide, including waterfowl production, one of the primary purposes of these lands. These results can be used to help guide FWS staff in future research and management decisions.

2.3.3. Future Research Needs

While this study looked at overall condition of PPR wetlands on FWS fee-title lands and where to expect significant differences in condition, there is still much more to be learned about wetland conditions in the area. This study opened opportunities for a significant amount of future research that may be important to understand how factors such as management and climate change are playing into wetland condition, how wetland conditions can be improved, how wetland management in the region can be improved, and how condition assessments can be improved.

For the purposes of this study, wetland sites were designated as being on native or reseeded grassland. While field crews were able to change the designation during site visits if substantial evidence showed the original designation appeared to be incorrect (e.g. designated as native prairie but had rock piles from previous cultivation), a majority of sites remained as they were originally designated. While sites have a reseeded or restored designation there was no information provided on when sites were reseeded or restored, and what species were initially reseeded. Studies show plant communities, species richness, and wetland conditions change significantly in the years following restoration depending on many factors such as years since restoration and species used in the reseeding mix (Galatowitsch and van der Valk 1996b;

Mullhouse and Galtowitsch 2003; Seabloom and van der Valk 2003a; Myla et al. 2008). If information could be obtained regarding when reseeding took place for fee-title lands where wetlands assessments were conducted and what seed mixes were used, analysis could possibly be done to see how wetland conditions change in reference to time since reseeding. It could also be determined which seed mixes accounted for the best and worst wetland conditions on reseeded fee-title lands. Continued monitoring of the more recent sites reseeded into diverse native mixes would also provide an opportunity to observe the progression of wetland conditions after reseeding.

Regarding how climate can affect wetlands, the 2021 field season, in which 100 condition assessments were conducted, was an abnormally dry year both before and during the growing season. Sites could be visited again in future years to compare assessment scores and condition categories with climate variations. This would not only provide insight on how conditions and plant communities appear to change with climate variation, but also would either help prove the validity of the IPCI or show a need for it to be recalibrated to account more for climate variation as proposed by Euliss and Mushet (2011). Continued monitoring would also show if the IPCI is still accurately following the disturbance gradient of the region it was designed with or if the metric value ranges need to be adjusted to meet current conditions and disturbance gradients; especially for temporary wetlands which appeared to be scoring lower due to their smaller size.

Future research could also look at the values or associated scores for each individual metric of the IPCI and NDRAM assessments. Each individual metric could be tested against variables such as native versus reseeded grassland, plant community composition, or cover of invasive species. Breaking assessments down to their individual metrics could provide insight into how more targeted, specific aspects of the wetlands are influenced by variables within and around the wetlands.

The FWS should also assess wetland conditions on tracts of land where landscape scale management programs like NPAM are being used to compare against wetland conditions where these programs are not actively being employed. NPAM uses fire, grazing, and rest periods to attempt to recreate natural disturbance patterns aimed at reducing invasive species cover and restoring grassland ecological integrity (Dixon et al. 2019). Since wetlands in these areas also evolved with these same disturbances it would be beneficial to determine how invasive cover and ecological integrity of wetlands on these lands are being affected by these management strategies in comparison to wetlands where they are not implemented.

Another likely need is to ensure plant community-based wetland assessments such as the FQI and IPCI are accurate which would involve reconvening to reassess the C-values associated with each species provided by TNGPFQAP (2001). Even though expert assigned C-values have been proven reliable for wetland condition assessments (Matthews et al. 2015), it has been 20 years since the panel last convened to assign species C-values and it is likely some species need to be reevaluated; many species and subspecies have also changed names, been separated, or joined together in that time. Given environmental changes over the last 20 years, an increased understanding of the roles some species play in the ecosystem, and an increased understanding of species' tolerance to disturbance there are likely species in need of updated C-values. This would increase the accuracy of wetland conditions assessments such as the FQI or IPCI.

2.3.4. Management Implications

The FWS has evolved their land management perspectives and goals in the PPR to focus more on biological diversity and ecological integrity (Dixon et al. 2019). They continue to acquire fee-title lands throughout the region, but now focus on restoring the land for their new goals rather than just for habitat of waterfowl. This study will help shed light on how to improve upon their current goals.

There was overall low diversity of plant species in wetlands on most reseeded grasslands even after management to improve the diversity and integrity of these areas. Many of these fee-title lands were acquired before present day management goals and were often restored solely for waterfowl habitat rather than diversity and ecological integrity (Dixon et al. 2019). Due to this, more emphasis should be put on initial restoration of recently acquired land and maintenance of diversity and integrity on lands which are already in good condition such as native prairie. In this study, wetlands in reseeded grassland contained high covers of invasive species while those in native grassland had higher cover of native species and accounted for a high percentage of overall native plant diversity, even though significantly fewer were surveyed than those in reseeded grassland. It would be easier to increase or maintain diversity of native species and stop invasive species from becoming dominant in areas where they are sparse or absent rather than attempting to completely change the plant community in areas where they have already taken over. Limiting the spread of these invasive species into native prairie areas which already have high diversity and ecological integrity should be of the utmost importance. This will help ensure the wetlands in these areas stay in good condition for the foreseeable future.

Commonly accepted ways of increasing diversity and ecological integrity include the use of grazing and fire (Dixon et al. 2019; Larson et al. 2020). Implementing these types of natural disturbance on native grassland and newly acquired land will help to increase or maintain native diversity and prevent invasive species from becoming dominant and outcompeting native species like we have seen on many of the older reseeded grasslands. Many of the reseeded grassland areas appear to be too dominated by invasive species to return to a point where they could resemble native grasslands. Unless groundbreaking invasive species management techniques are discovered to eliminate these species from areas they are dominating, it makes more sense to use limited resources on areas where they will make more of a difference. This is not to say these invasive dominated reseeded grasslands should not be managed at all, but rather managed to maintain ecological goals requiring less focus and resources, allowing more of those resources to be used on native prairies and newly acquired lands where they will have a greater impact.

Future acquisition of FWS fee-title lands should focus on native prairie areas, lands adjacent to native prairie, and large tracts of land which encompass significant portions of wetland complexes. Native prairie areas still can have high diversity and ecological integrity within the region. Acquiring these lands to ensure they are being managed properly using grazing and fire regimes should be important in upcoming years when climate change and anthropogenic disturbance continue to alter the land. Dispersal limitation is a major reason why native species diversity is so low in reseeded grassland (Mullhouse and Galatowitsch 2003; Seabloom and van der Valk 2003b). This was evident in this study as well. Acquiring lands adjacent to native prairie, even if it is previously cultivated, will allow for a greater chance for natural seed dispersal from the nearby native grasslands and wetlands. This will result in a greater increase in native species diversity for these lands than it would for lands isolated and surrounded by predominantly agriculture. Isolated lands need to be acquired as well to ensure there is wildlife habitat in many different areas of the PPR but is not as important for achieving the overall goals of diversity and ecological integrity. Lastly, the FWS should also focus on acquiring larger tracts of land whenever possible rather than many small, isolated lands. This means restoration of a greater number and area of wetlands which will increase the chance of native species seed dispersal throughout the wetland complexes in the years following restoration and increase the likelihood these wetlands are able to perform necessary ecosystem services.

3. FWS FEE-TITLE LAND WETLAND PLANT COMMUNITY COMPOSITION DRIVERS

3.1. Methods

3.1.1. Plant Identification and Taxonomy

Both primary and secondary species were identified down to the species level whenever possible by field crews conducting vegetation surveys following the IPCI method (DeKeyser et al. 2003; Hargiss et al. 2008). Field crews were trained in plant identification prior to each field season. Any plants which field crews were not able to be identify down to the species level were identified to the genus level, but could not be used for calculating IPCI metric scores or for plant community composition analysis. In the case a plant species could not be identified in the field but was in good enough condition to be identified by someone with more expertise, an individual plant was collected and later identified in the North Dakota State University Herbarium. For data collection and data analysis six letter acronyms were used for each species. Each acronym was comprised of the first three letters of the genus names and first three letters of the species name. The only exceptions to this were the use of different acronyms when the six-letter acronym matches another exactly and for species in the Carex genus which were designated by Cx followed by the first four letters of the species name. Plant scientific names, common names, origin (Native or Introduced), and life-history guild (Annual, Biennial, Perennial) were taken from the U.S. Department of Agriculture’s PLANTS Database (USDA NRCS 2021). TNGPFQAP (2001) was used to determine plant C-values (0-10). A complete species list of all primary and secondary species identified was compiled along with previously mentioned characteristics and can be found in Appendix A.

3.1.2. Species Frequency and Relative Abundance

Species cover and abundance data were collected using the IPCI developed for the PPR by DeKeyser et al. (2003) and Hargiss et al. (2008). Using the IPCI primary and secondary species data were collected for each wetland. Using this, a list of total species present for each wetland was created. The complete species list for all wetlands was combined to determine all species observed for the 200 wetlands surveyed. This data was also used to create a cumulative list of species presence throughout the region. This includes the total amount of wetlands for which each species was present; the most commonly found native and introduced species were recorded and organized in order of species frequency.

Aerial cover of primary species was determined using the quadrat method developed for IPCI surveys (DeKeyser et al 2003; Hargiss et al. 2008). Primary species cover was estimated using 1-meter by 1-meter quadrats where all species within the quadrat were identified to the species level when possible, and genus level when not. Cover of each species within the quadrat were estimated to the nearest 1.0%. For statistical purposes any species with less than 1.0% cover estimation was assigned as 0.1% cover. Primary species data was collected for all 200 wetlands. Cover estimate data was kept specific to each zone rather than attempting to combine each vegetation zone because vegetation zones are viewed as different plant communities within each wetland. Species-cover percentage estimates for each vegetation zone of each individual wetland was averaged by taking the sum of species-cover estimates in all quadrats and dividing it by the number of quadrats used in the specific vegetation zone (8 for low-prairie, 7 for wet-meadow, and 5 for shallow-marsh). Average cover for each species was summed to give the average cumulative vegetation cover of each wetland. Given the cumulative vegetation cover and the species-specific cover, the relative cover of each species was determined by dividing the average species-cover by the cumulative vegetation cover. The relative cover gives the percentage of present vegetation for which each species makes up and takes aerial cover of litter and bare ground out of the equation.

3.1.3. Data Analysis

Nonmetric multi-dimensional scaling (NMS) was conducted in PC-ORD version 7 software (McCune and Mefford 1999; McCune et al. 2002) to examine seasonal and temporary wetland plant communities. To ensure data was not skewed by overabundance of zeros, datasets were trimmed to exclude species which did not occur in at least 5% of the wetland surveyed (n=7 for seasonal wetlands and n=3 for temporary wetlands). Plant species trimmed and those retained for NMS can be seen in Appendix C. Relative cover values were modified using the arcsine square root transformation.

Indication of relations among wetland sites in species space were made using NMS ordination. NMS ordination followed the methods of Smith et al. (2016). The NMS ordination utilized a relative Sørenson distance measure, a random starting point, 50 runs with real data, 250 runs with randomized data, a step length of 0.20, and a stability criterion of 0.0005. Species significantly correlated with the NMS axes, or those possessing a Pearson correlation coefficient with an absolute value ≥ 0.4, were considered significant drivers of the axis. However, species not significantly correlated were still examined further to help determine drivers of each axis.

Comparisons among wetland plant communities for sites and habitat characteristics were made using multi-response permutation procedure (MRPP) following Smith et al. (2016). MRPP with the relative Sørenson distance measure was used to compare wetland plant communities among pothole classes and zones. Pairwise comparisons for each wetland zone were done between grassland type (native versus reseeded).

3.2. Results

3.2.1. Species Frequency

In total the 200 IPCI assessments resulted in 348 different species from 207 different genera observed during field data collection. Of those 348 species 60 were annuals, 13 were biennials, and 275 were perennials with 285 of those species being native and 63 being introduced. There were 25 species present in at least 50% of wetlands (n=100). Species present in at least 50% of wetlands can be found in Table 3.1. There were 8 species present in at least 75% of wetlands (n=150). Those 8 species were *Cirsium arvense* (Canada thistle), *Poa pratensis* (Kentucky bluegrass), *Bromus inermis* (smooth brome), *Sonchus arvensis* (field sowthistle), *Symphyotrichum lanceolatum* (white panicle aster), *Solidago canadensis* (Canada goldenrod), *Rumex crispus* (curly dock), and *Polygonum amphibium var. stipulaceum* (swamp smartweed). There were 2 species present in 100% of wetlands surveyed (n=200): *Cirsium arvense* (Canada thistle), and *Poa pratensis* (Kentucky bluegrass). Both are introduced species, with the next 2 most frequently occurring species and 9 of the 25 most frequently occurring species also being introduced. Other species of interest are *Phalaris arundinacea* (reed canarygrass) which was present in 72.5% of wetlands (n=145) and *Typha x glauca* (hybrid cattail) which was present in 55% of wetlands (n=110).

Table 3.1. Species present in at least 50% of IPCI assessments.

Species	C[a]	n[b]
Cirsium arvense	*	200
Poa pratensis	*	200
Bromus inermis	*	195
Sonchus arvensis	*	184
Symphyotrichum lanceolatum	3	174
Solidago canadensis	1	167
Rumex crispus	*	150
Polygonum amphibium var. stipulaceum	0	150
Carex pellita	4	149
Asclepias speciosa	4	149
Phalaris arundinacea	0	145
Elymus repens	*	145
Symphyotrichum ericoides	2	130
Symphoricarpos occidentalis	3	123
Spartina pectinata	5	122
Rosa woodsii	5	117
Glycyrrhiza lepidota	2	114
Carex atherodes	4	111
Typha x glauca	*	110
Eleocharis palustris	4	110
Stachys pilosa	3	109
Melilotus officinalis	*	109
Hordeum jubatum	0	108
Anemone canadensis	4	107
Artemisia absinthium	*	103

[a]Coefficient of Conservatism (NGPFQAP 2001)
[b]Number of wetlands in which the species was present
*Introduced species are not assigned a coefficient of conservatism

3.2.2. Temporary Wetlands

Low prairie relative cover data for temporary wetlands was trimmed to exclude any species which did not appear in at least 5% of the 59 temporary wetlands surveyed (n=3). This function deleted 63 species columns which trimmed the dataset from 126 species down to 63 species. MRPP analysis of the trimmed data did not show a significant difference ($p<0.05$) in the low prairie plant communities between grassland types (native versus reseeded).

Non-metric multidimensional scaling analysis of the low prairie zone produced a final solution with 3 dimensions or 3 axes. The 3-dimensional solution had a final stress of 14.07995, 62 iterations for the final solution, and a final instability of 0.00047. All 3 dimensions, or axes, are necessary to explain the variation in the low prairie plant communities. The 3 axes cumulatively accounted for 84.6% of variation in the low prairie plant communities. Axis 1 accounted for 41.1% of variation, axis 2 accounted for 22.5%, and axis 3 accounted for 20.9%.

Axis 1, which accounted for the most variation of the 3 axes produced by the NMS ordination, had a significant correlation (Pearson correlation ≥ 0.40) with 4 species. *Elymus repens* (quackgrass) had the strongest positive correlation with axis 1 (0.516) (Table 3.2). *Bromus inermis* (smooth brome) was the lone species negatively correlated with axis 1 (-0.912) (Figure 3.1). Axis 2 accounted for the second most variation of the 3 axes produced by the NMS ordination and had a significant correlation (Pearson correlation ≥ 0.40) with 5 species. *Poa pratensis* (Kentucky bluegrass) had the strongest positive correlation with axis 2 (0.610) (Figure 3.2). *Symphoricarpos occidentalis* (western snowberry) had the strongest negative correlation with axis 2 (-0.509). Axis 3 accounted for the least variation of the 3 axes produced by the NMS ordination and had a significant correlation (Pearson correlation ≥ 0.40) with 8 species. *Poa pratensis* (Kentucky bluegrass) had the strongest positive correlation with axis 3 (0.558). *Ratibida columnifera* (upright prairie coneflower) had the strongest negative correlation with axis 3 (-0.474).

Table 3.2. Plant species with a correlation (Person correlation ≥ 0.40) between relative cover and non-metric multidimensional scaling ordination axes for the low prairie zone of temporary wetlands.

Species	C[a]	Axis 1[b]	Axis 2[b]	Axis 3[b]
Achillea millefolium	3			-0.410
Anemone canadensis	4		-0.477	
Artemisia ludoviciana	3		-0.504	0.434
Bromus inermis	*	-0.912		
Elaeagnus commutata	5			0.417
Elymus repens	*	0.516		
Galium boreale	4			0.422
Pascopyrum smithii	4		0.441	
Poa pratensis	*		0.610	0.558
Ratibida columnifera	3			-0.474
Rosa woodsii	5			0.402
Solidago canadensis	1	0.402		
Symphoricarpos occidentalis	3		-0.509	0.529
Symphyotrichum lanceolatum	3	0.446		

[a]Coefficient of Conservatism (NGPFQAP 2001)
[b]Pearson correlation with NMS axes
*Introduced species are not assigned a coefficient of conservatism

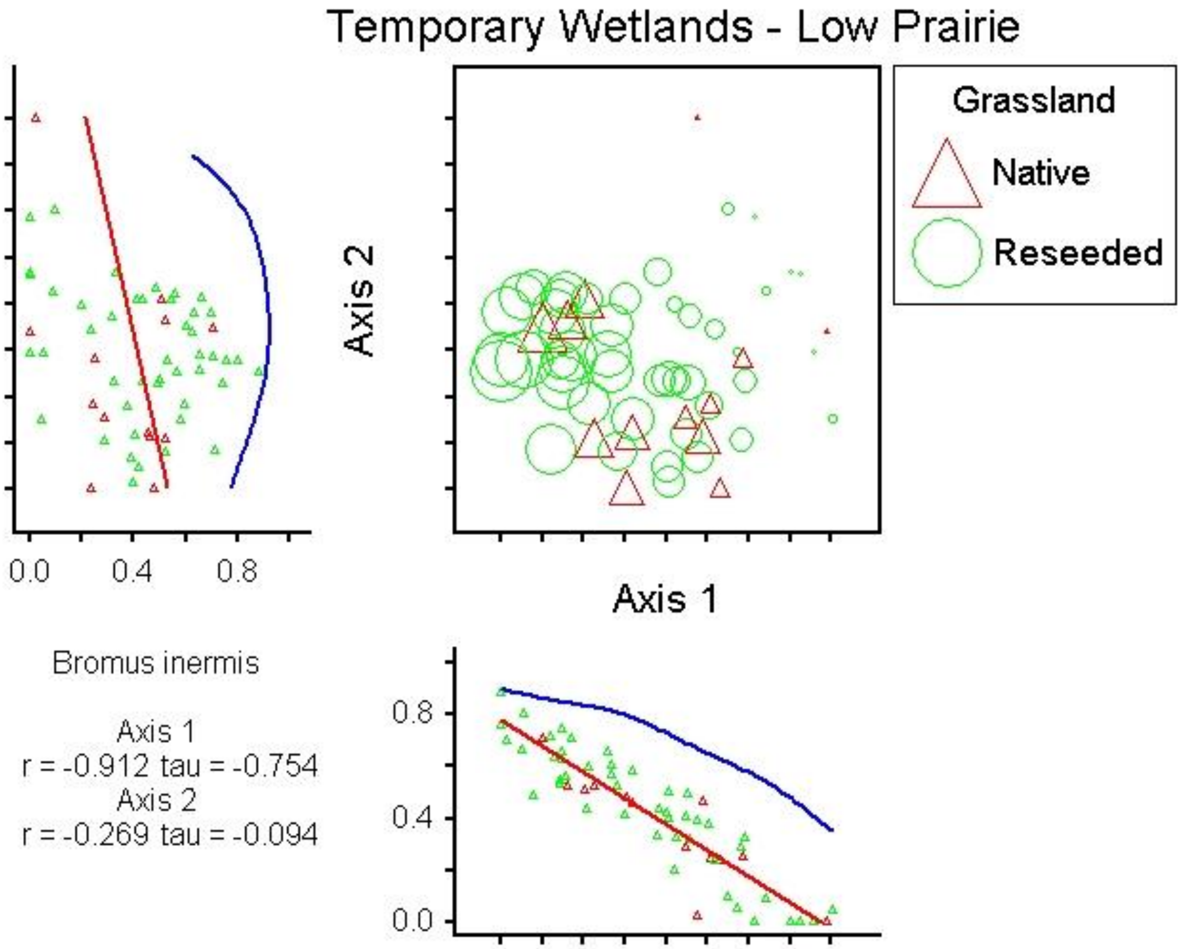

Figure 3.1. Correlation of *Bromus inermis* for axes 1 and 2 from NMS ordination of the low prairie zone of temporary wetlands. Figure Note: Each individual circle and triangle in the central graph represents a temporary wetland site. Shape size is scaled by relative cover of *Bromus inermis* at the wetland site.

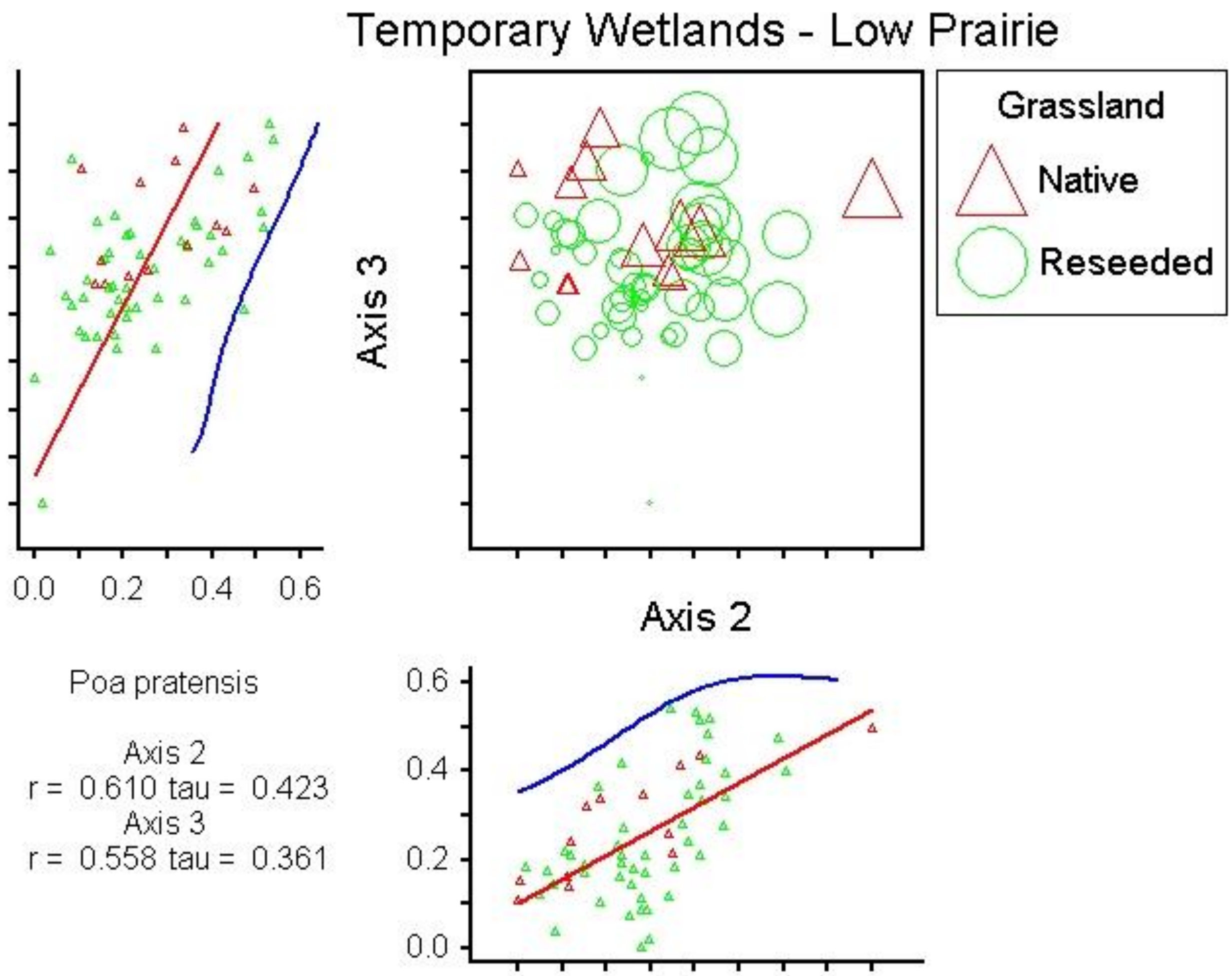

Figure 3.2. Correlation of *Poa pratensis* for axes 2 and 3 from NMS ordination of the low prairie zone of temporary wetlands. Figure Note: Each individual circle and triangle in the central graph represents a temporary wetland site. Shape size is scaled by relative cover of *Poa pratensis* in the low prairie zone at the wetland site.

Wet meadow relative cover data for temporary wetlands was trimmed to exclude any species which did not appear in at least 5% of the 59 temporary wetlands surveyed (n=3). This function deleted 69 species columns which trimmed the dataset from 112 species down to 43 species. MRPP analysis of the trimmed data showed a significant difference ($p<0.05$) in the wet meadow plant communities between grassland types (native versus reseeded).

Non-metric multidimensional scaling analysis of the low prairie zone produced a final solution with 3 dimensions or 3 axes. The 3-dimensional solution had a final stress of 16.8367, 65 iterations for the final solution, and a final instability of 0.00048. All 3 dimensions, or axes, are necessary to explain the variation in the low prairie plant communities. The 3 axes cumulatively accounted for 73.1% of variation in the low prairie plant communities. Axis 1 accounted for 29.2% of variation, axis 2 accounted for 26.2%, and axis 3 accounted for 17.7%.

Axis 1, which accounted for the most variation of the 3 axes produced by the NMS ordination, had a significant correlation (Pearson correlation ≥ 0.40) with 2 species. *Elymus repens* (quackgrass) was the lone species positively correlated with axis 1 (0.521) (Table 3.3). *Phalaris arundinacea* (reed canarygrass) was the lone species negatively correlated with axis 1 (-0.919) (Figure 3.3). Axis 2 accounted for the second most variation of the 3 axes produced by the NMS ordination and had a significant correlation (Pearson correlation ≥ 0.40) with 4 species. *Hordeum jubatum* (foxtail barley) had the strongest positive correlation with axis 2 (0.635). *Polygonum amphibium var. stipulaceum* (swamp smartweed) was the lone species negatively correlated with axis 2 (-0.611). Axis 3 accounted for the least variation of the 3 axes produced by the NMS ordination and had a significant correlation (Pearson correlation ≥ 0.40) with 2 species. Both species were negatively correlated to axis 3. *Carex laeviconica* (smoothcone sedge) had the strongest negative correlation (-0.702).

Table 3.3. Plant species with a correlation (Person correlation ≥ 0.40) between relative cover and non-metric multidimensional scaling ordination axes for the wet meadow zone of temporary wetlands.

Species	C[a]	Axis 1[b]	Axis 2[b]	Axis 3[b]
Calamagrostis canadensis	5			-0.521
Carex laeviconica	6			-0.702
Eleocharis palustris	4		0.516	
Elymus repens	*	0.521		
Hordeum jubatum	0		0.635	
Phalaris arundinacea	0	-0.919		
Polygonum amphibium var. stipulaceum	0		-0.611	
Rumex crispus	*		0.600	

[a]Coefficient of Conservatism (NGPFQAP 2001)
[b]Pearson correlation with NMS axes
*Introduced species are not assigned a coefficient of conservatism

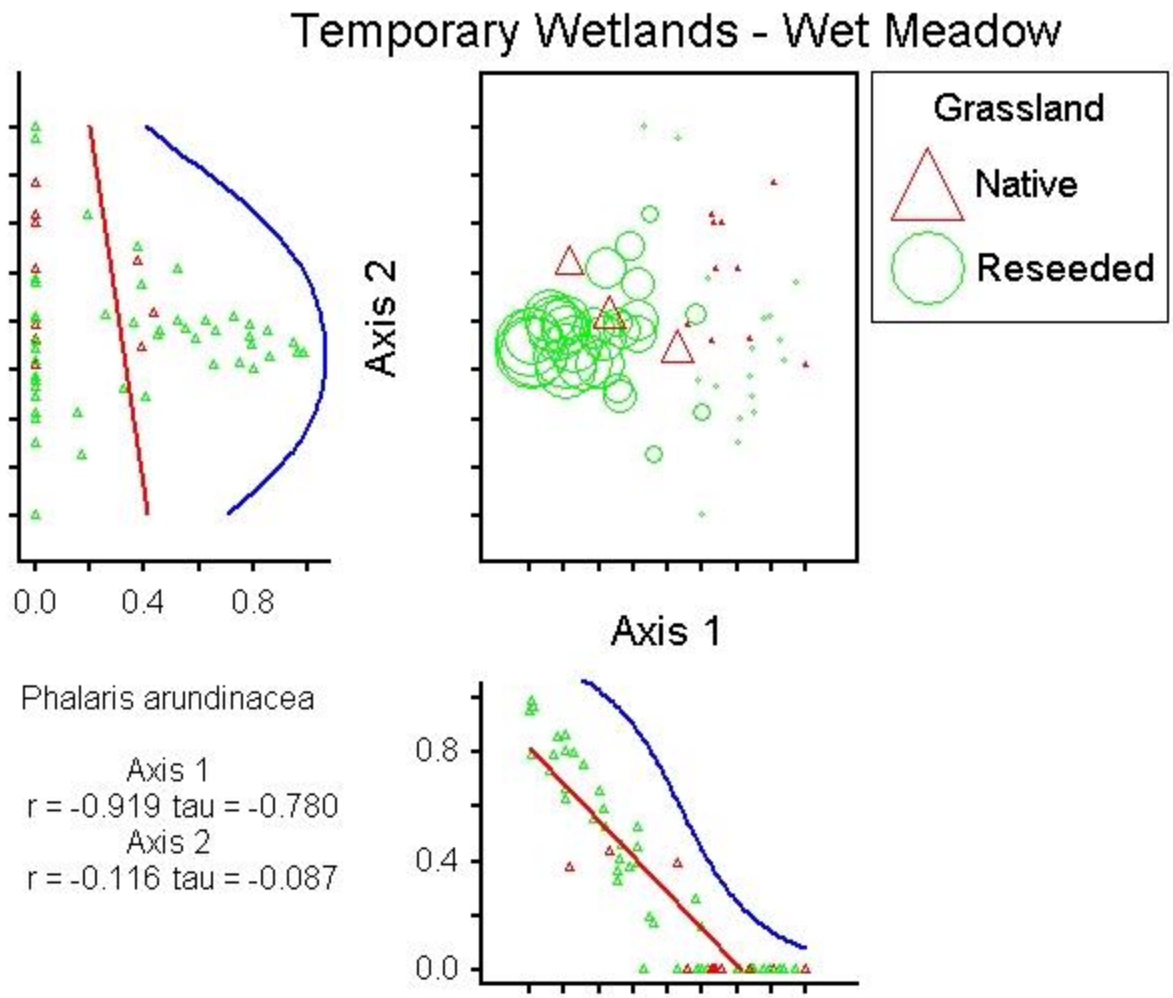

Figure 3.3. Correlation of *Phalaris arundinacea* for axes 1 and 2 from NMS ordination of the wet meadow zone of temporary wetlands. Figure Note: Each individual circle and triangle in the central graph represents a temporary wetland site. Shape size is scaled by relative cover of *Phalaris arundinacea* in the wet meadow zone at the wetland site.

3.2.3. Seasonal Wetlands

Low prairie relative cover data for temporary wetlands was trimmed to exclude any species which did not appear in at least 5% of the 141 seasonal wetlands surveyed (n=7). This function deleted 120 species columns which trimmed the dataset from 183 species down to 63 species. MRPP analysis of the trimmed data showed a significant difference ($p<0.05$) in the low prairie plant communities between grassland types (native versus reseeded).

Non-metric multidimensional scaling analysis of the low prairie zone produced a final solution with 3 dimensions or 3 axes. The 3-dimensional solution had a final stress of 17.1565, 77 iterations for the final solution, and a final instability of 0.0004. All 3 dimensions, or axes, are necessary to explain the variation in the low prairie plant communities. The 3 axes cumulatively accounted for 80.1% of variation in the low prairie plant communities. Axis 1 accounted for 39.1% of variation, axis 2 accounted for 22.5%, and axis 3 accounted for 18.5%.

Axis 1, which accounted for the most variation of the 3 axes produced by the NMS ordination, had a significant correlation (Pearson correlation ≥ 0.40) with 2 species. *Elymus repens* (quackgrass) was the lone species positively correlated with axis 1 (0.506) (Table 3.4). *Bromus inermis* (smooth brome) was the lone species negatively correlated with axis 1 (-0.895) (Figure 3.4). Axis 2 accounted for the second most variation of the 3 axes produced by the NMS ordination and had a significant correlation (Pearson correlation ≥ 0.40) with 9 species. *Poa pratensis* (Kentucky bluegrass) had the strongest positive correlation with axis 2 (0.594) (Figure 3.5). *Melilotus officinalis* (sweetclover) was the lone species negatively correlated with axis 2 (-0.509). Axis 3 accounted for the least variation of the 3 axes produced by the NMS ordination and had a significant correlation (Pearson correlation ≥ 0.40) with 2 species. *Solidago canadensis* (Canada goldenrod) was the lone species positively correlated with axis 3 (0.428). *Poa pratensis* (Kentucky bluegrass) was the lone species negatively correlated with axis 3 (-0.567).

Table 3.4. Plant species with a correlation (Person correlation ≥ 0.40) between relative cover and non-metric multidimensional scaling ordination axes for the low prairie zone of seasonal wetlands.

Species	C[a]	Axis 1[b]	Axis 2[b]	Axis 3[b]
Anemone canadensis	4		0.494	
Artemisia ludoviciana	3		0.424	
Bromus inermis	*	-0.895		
Elaeagnus commutata	5		0.463	
Elymus repens	*	0.506		
Galium boreale	4		0.487	
Helianthus pauciflorus	8		0.430	
Melilotus officinalis	*		-0.409	
Poa pratensis	*		0.594	-0.567
Rosa arkansana	5		0.457	
Solidago canadensis	1			0.428
Symphoricarpos occidentalis	3		0.560	

[a]Coefficient of Conservatism (NGPFQAP 2001)
[b]Pearson correlation with NMS axes
*Introduced species are not assigned a coefficient of conservatism

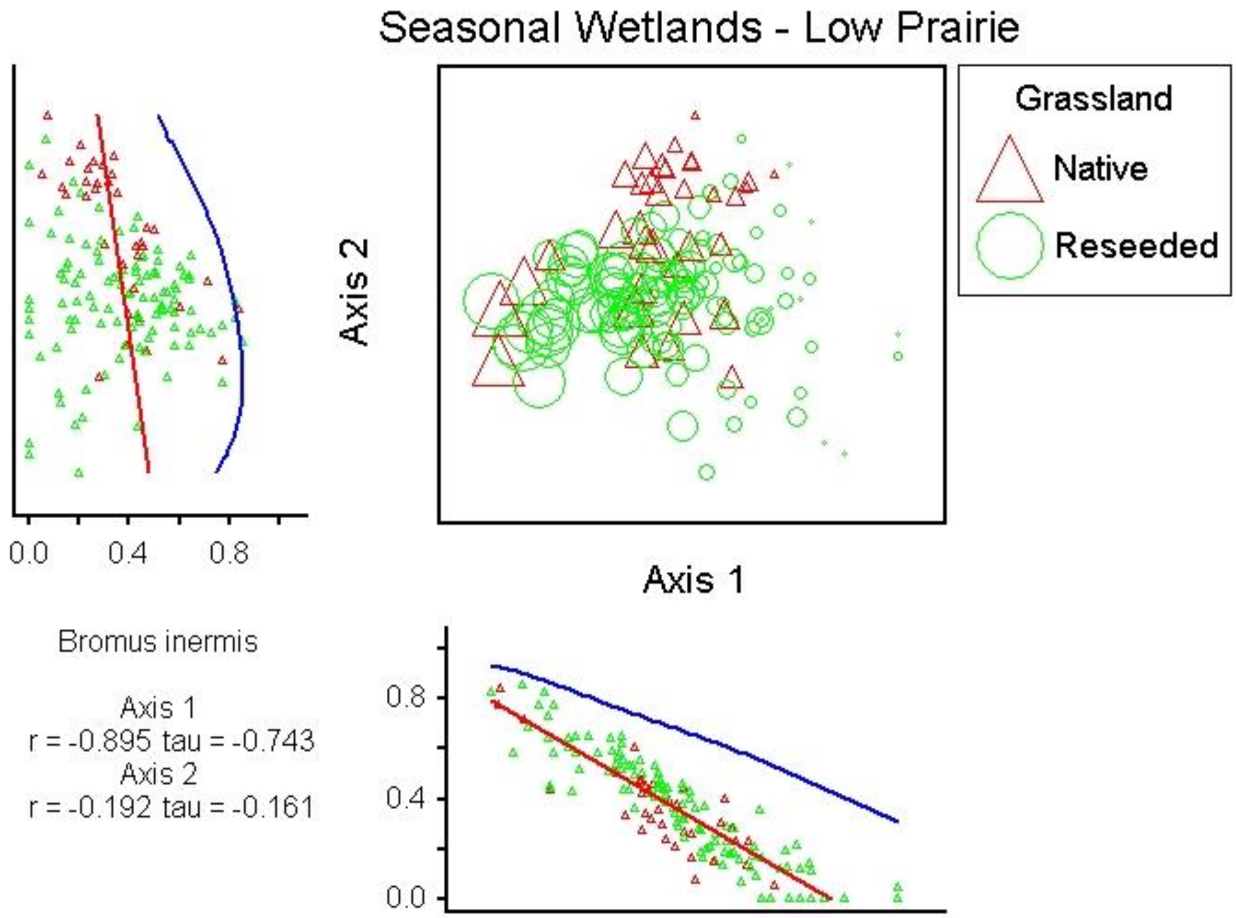

Figure 3.4. Correlation of *Bromus inermis* for axes 1 and 2 from NMS ordination of the low prairie zone of seasonal wetlands. Figure Note: Each individual circle and triangle in the central graph represents a seasonal wetland site. Shape size is scaled by relative cover of *Bromus inermis* at the wetland site.

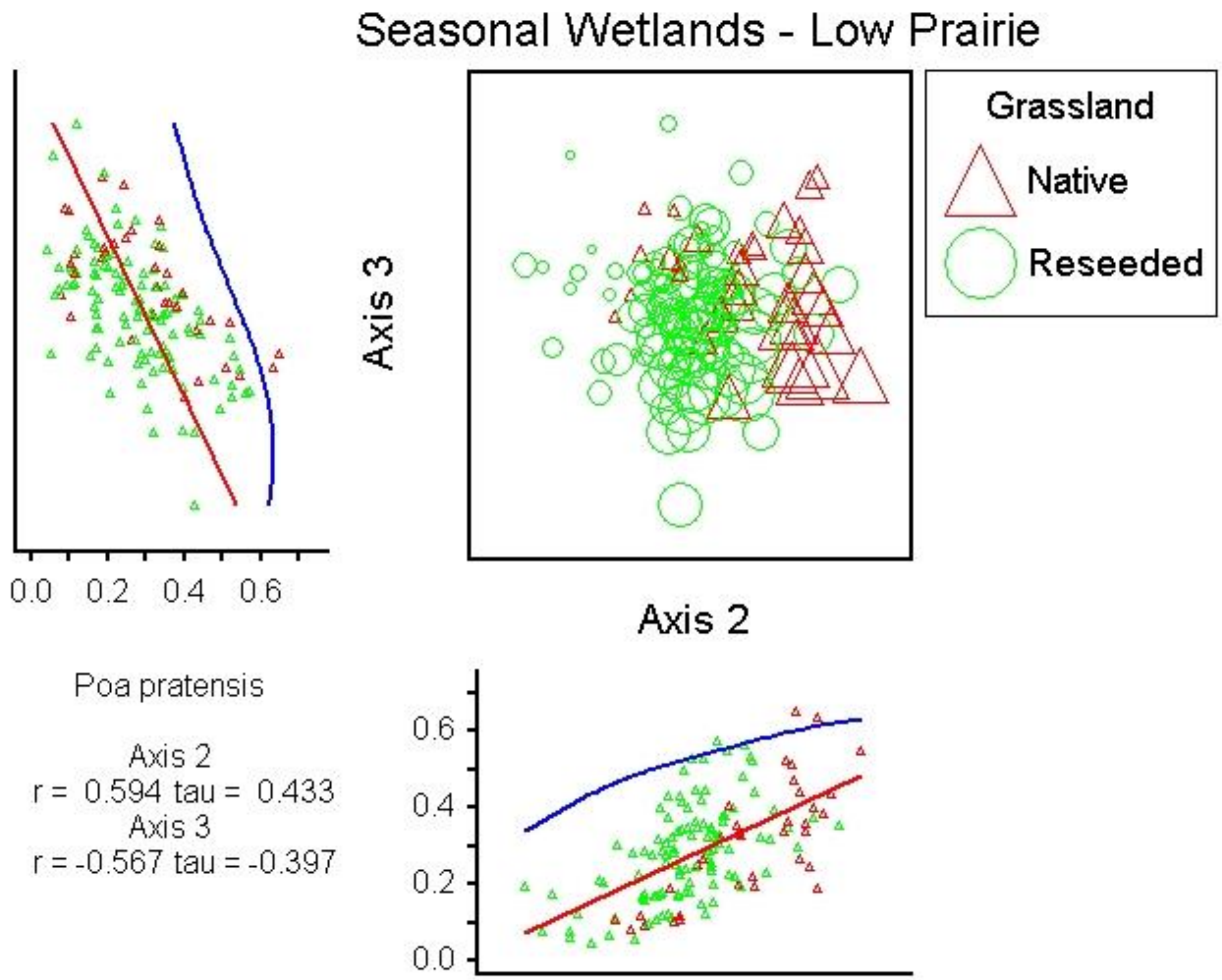

Figure 3.5. Correlation of *Poa pratensis* for axes 2 and 3 from NMS ordination of the low prairie zone of seasonal wetlands. Figure Note: Each individual circle and triangle in the central graph represents a seasonal wetland site. Shape size is scaled by relative cover of *Poa pratensis* in the low prairie zone at the wetland site.

Wet meadow relative cover data for seasonal wetlands was trimmed to exclude any species which did not appear in at least 5% of the 141 seasonal wetlands surveyed (n=7). This function deleted 106 species columns which trimmed the dataset from 162 species down to 56 species. MRPP analysis of the trimmed data showed a significant difference ($p<0.05$) in the wet meadow plant communities between grassland types (native versus reseeded).

Non-metric multidimensional scaling analysis of the low prairie zone produced a final solution with 3 dimensions or 3 axes. The 3-dimensional solution had a final stress of 17.5803, 73 iterations for the final solution, and a final instability of 0.00048. All 3 dimensions, or axes, are necessary to explain the variation in the low prairie plant communities. The 3 axes cumulatively accounted for 75.7% of variation in the wet meadow plant communities. Axis 1 accounted for 41.7% of variation, axis 2 accounted for 20.3%, and axis 3 accounted for 13.7%.

Axis 1, which accounted for the most variation of the 3 axes produced by the NMS ordination, had a significant correlation (Pearson correlation ≥ 0.40) with 2 species. *Spartina pectinata* (prairie cordgrass) was the lone species positively correlated with axis 1 (0.509) (Table 3.5). *Phalaris arundinacea* (reed canarygrass) was the lone species negatively correlated with axis 1 (-0.933) (Figure 3.6). Axis 2 accounted for the second most variation of the 3 axes produced by the NMS ordination and had a significant correlation (Pearson correlation ≥ 0.40) with 2 species. *Calamagrostis canadensis* (bluejoint) was the lone species positively correlated with axis 2 (0.537). *Spartina pectinata* (prairie cordgrass) was the lone species negatively correlated with axis 2 (-0.651). Axis 3 accounted for the least variation of the 3 axes produced by the NMS ordination and had a significant correlation (Pearson correlation ≥ 0.40) with 2 species. *Carex pellita* (wooly sedge) was the lone species positively correlated with axis 3 (0.709). *Polygonum amphibium var. stipulaceum* (swamp smartweed) was the lone species negatively correlated with axis 3 (-0.569).

Table 3.5. Plant species with a correlation (Person correlation ≥ 0.40) between relative cover and non-metric multidimensional scaling ordination axes for the wet meadow zone of seasonal wetlands.

Species	C[a]	Axis 1[b]	Axis 2[b]	Axis 3[b]
Calamagrostis canadensis	5		0.537	
Carex pellita	4			0.709
Phalaris arundinacea	0	-0.933		
Polygonum amphibium var. stipulaceum	0			-0.569
Spartina pectinata	5	0.509	-0.651	

[a]Coefficient of Conservatism (NGPFQAP 2001)
[b]Pearson correlation with NMS axes
*Introduced species are not assigned a coefficient of conservatism

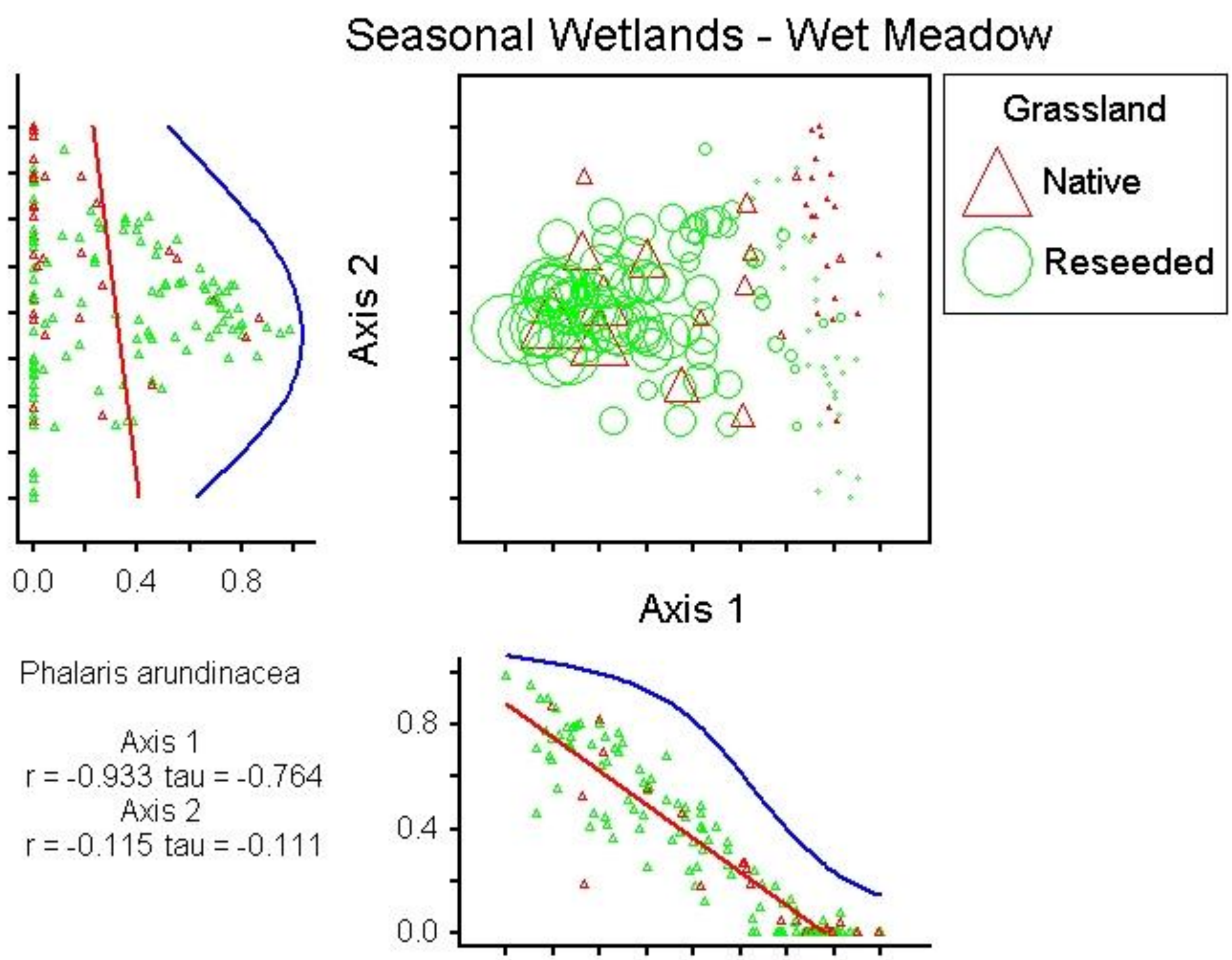

Figure 3.6. Correlation of *Phalaris arundinacea* for axes 1 and 2 from NMS ordination of the wet meadow zone of seasonal wetlands. Figure Note: Each individual circle and triangle in the central graph represents a seasonal wetland site. Shape size is scaled by relative cover of *Phalaris arundinacea* in the wet meadow zone at the wetland site.

Shallow marsh relative cover data was trimmed to exclude any species which did not appear in at least 5% of the 141 seasonal wetlands surveyed (n=7). This function deleted 77 species columns which trimmed the dataset from 105 species down to 28 species. MRPP analysis of the trimmed data showed a significant difference ($p<0.05$) in the shallow marsh zone between grassland types (native versus reseeded).

Non-metric multidimensional scaling analysis of the shallow marsh zones produced a final solution with 3 dimensions or 3 axes. The 3-dimensional solution had a final stress of 13.70462, 95 iterations for the final solution, and a final instability of 0.0005. All 3 dimensions, or axes, are necessary to explain the variation in the low prairie plant communities. The 3 axes cumulatively accounted for 82.9% of variation in the wet meadow plant communities. Axis 1 accounted for 42.7% of variation, axis 2 accounted for 25.2%, and axis 3 accounted for 15%.

Axis 1, which accounted for the most variation of the 2 axes produced by the NMS ordination, had a significant correlation (Pearson correlation ≥ 0.40) with 4 species. *Polygonum amphibium var. stipulaceum* (swamp smartweed) had the strongest positive correlation with axis 1 (0.594) (Table 3.6). *Typha x glauca* (hybrid cattail) was the lone species negatively correlated with axis 1 (-0.884) (Figure 3.7). Axis 2 accounted for the second most variation of the 3 axes produced by the NMS ordination and had a significant correlation (Pearson correlation ≥ 0.40) with 2 species. *Carex atherodes* (wheat sedge) was the lone species positively correlated with axis 2 (0.617). *Lemna turionifera* (turion duckweed) was the lone species negatively correlated with axis 2 (-0.526). Axis 3 accounted for the least variation of the 3 axes produced by the NMS ordination and had a significant correlation (Pearson correlation ≥ 0.40) with 2 species. Both species were positively correlated to axis 3. *Phalaris arundinacea* (reed canarygrass) had the strongest positive correlation with axis 3 (0.545).

Table 3.6. Plant species with a correlation (Person correlation ≥ 0.40) between relative cover and non-metric multidimensional scaling ordination axes for the shallow marsh zone of seasonal wetlands.

Species	C[a]	Axis 1[b]	Axis 2[b]	Axis 3[b]
Carex atherodes	4	0.511	0.617	
Eleocharis palustris	4			0.417
Lemna turionifera	1		-0.526	
Phalaris arundinacea	0	0.412		0.545
Polygonum amphibium var. stipulaceum	0	0.594		
Typha x glauca	*	-0.884		

[a]Coefficient of Conservatism (NGPFQAP 2001)
[b]Pearson correlation with NMS axes
*Introduced species are not assigned a coefficient of conservatism

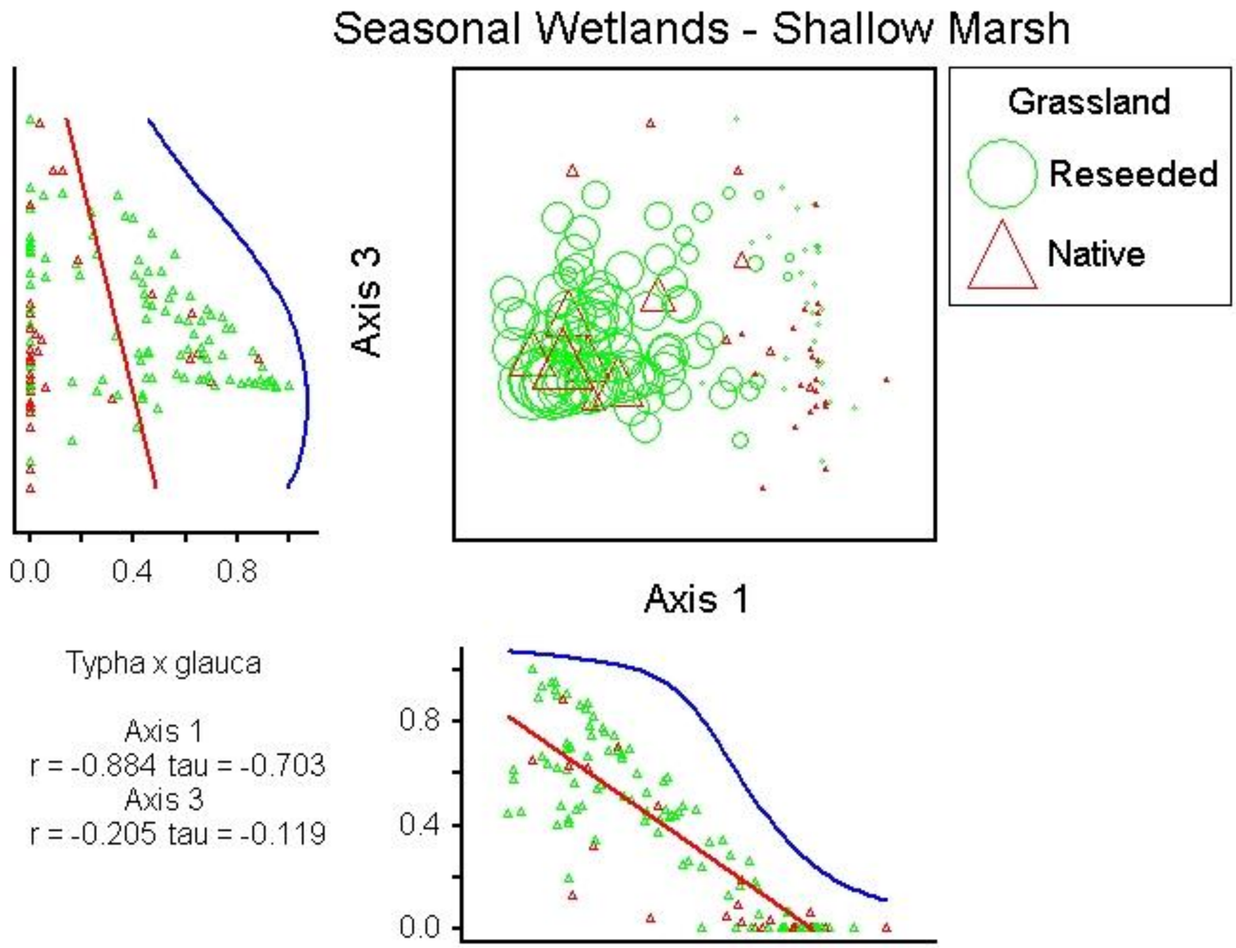

Figure 3.7. Correlation of *Typha x glauca* for axes 1 and 3 from NMS ordination of the shallow marsh zone of seasonal wetlands. Figure Note: Each individual circle and triangle in the central graph represents a seasonal wetland site. Shape size is scaled by relative cover of *Typha x glauca* in the shallow marsh zone at the wetland site.

3.3. Discussion

3.3.1. Species Frequency

Over the course of this study 348 different species were identified using the IPCI assessment method. However, many of those species occurred infrequently. Of the 348 species, only 25 appeared in 50% or more of the 200 wetland assessments. Even when considering species occurring in 25% or more of wetland assessments there were still only 47 species which occurred that frequently and 301 species found in less than 25% of wetlands assessed. This is primarily due to the site selection containing only a small sample of both pristine wetlands in native grassland and wetlands with very different conditions from current typical PPR wetlands (e.g. forested wetlands), which together accounted for a high percentage of the native species diversity observed. These results line up with previous research showing a high percentage of total species diversity can be found at very few of the overall sites and that certain plant communities and species within wetlands are highly sensitive to disturbance and often underrepresented in restored wetlands (Galatowitsch and van der Valk 1996b; Galatowitsch et al. 2000; Mullhouse and Galatowitsch 2003; Smith et al. 2016).

Most of the frequently occurring species are those found primarily in the low prairie or wet meadow zones seeing as every wetland assessed had those two zones but not every wetland had a shallow marsh zone. Shallow marsh species were likely underrepresented for species frequency. However, some species primarily found in the shallow marsh, such as *Typha x glauca* (hybrid cattail), were still among the most frequently occurring species. These species would likely have a higher frequency if all assessments were done on seasonal wetlands versus both seasonal and temporary.

Looking at Table 3.1, the four most frequently occurring species were all highly invasive introduced species commonly found in the low prairie areas of wetlands (Steward and Kantrud 1971; TNGPFQAP 2001). All four of those species were found in at least 90% of wetland assessments, with two (*Poa pratensis* and *Cirsium arvense*) being found in 100% of wetland assessments. Not only were these species frequently occurring, but they often had high relative cover as well. This mean some invasive species are now being frequently found in native prairie areas, not just highly disturbed or reseeded areas.

Native species did account for 16 of the 25 most frequently occurring species but most of those native species were ones that can handle fairly high disturbance levels. Looking at Table

3.1, no species found in at least 50% of wetland assessments had a C-values of greater than 5. Of the species found in 25% or more of wetland assessments only 2 species had a C-value of greater than 5: *Carex laeviconica* (smoothcone sedge) and *Helianthus pauciflorus* (stiff sunflower). However, 104 of the 348 species observed over the course of the study have a C-value of greater than 5. This reaffirms that much of the diversity of desired native species is being found only in the best condition wetlands and wetlands in average to poor conditions are supporting a low amount of the overall diversity of native plant species in the region. This also means many of the wetlands in the region experience too much disturbance or have too high of invasive species cover to support a high diversity of desired native plant species.

3.3.2. Temporary Wetlands

Non-metric multidimensional scaling (NMS) ordination produced 3 axes for both the low prairie and wet meadows zones. The amount of variation explained by each axes follows numerical order (1-3). Each axis is used to explain the variation among plant communities seen in the 59 temporary wetlands assessed. The axis explaining the most variation for each zone appeared to be controlled by cover of the most dominant invasive species.

Axis 1 had three positive species correlations and several others close to meeting the Pearson correlation (≥ 0.40) needed to be considered driving species, most of which were desired native species. Only *Bromus inermis* (smooth brome) had a significant negative correlation with axis 1 (-0.912). Grant et al. (2020b) showed a high frequency and cover of *Bromus inermis* being most prevalent in areas of low native species cover and vice versa in the same study area. This also confirms results of other previous studies looking at native versus invasive frequency and cover (Murphy and Grant 2005; Grant et al. 2009). *Bromus inermis* was present as a primary species in 54 of 59 temporary wetlands low prairie zones: often approaching 50-100% of relative cover, especially in reseeded grasslands. The correlation between *Bromus inermis* and axis 1 is evident in Figure 3.1. Sites with a high relative cover of *Bromus inermis* appear on the negative end of axis 1 while those with a low relative cover appear near the positive end. It appeared to be outcompeting all other species in the low prairie zone, invasive species included. It is known to dominate and spread, and to restrict and alter growth of other species (Dillemuth et al. 2009). This mechanism of invasion allows it to completely take over entire areas of disturbed grassland and edge out any remaining pockets of higher diversity.

The trend of axis 2 is not as obvious as axis 1. *Poa pratensis* (Kentucky bluegrass) has the highest correlation with axis 2 (0.610) and appears to be the major driving species. The trend between *Poa pratensis* cover and axis 2 can be seen in Figure 3.2 where sites with higher cover appear on the positive end and sites with lower cover appear on the negative end. Other species positively correlated to axis 2 are mostly those which can handle higher levels of disturbance frequently occurring when *Bromus inermis* cover is relatively low. Most species negatively correlated to axis 2 are native forbs and shrubs commonly found throughout the study region regardless of whether the site is in native or reseeded grassland.

Axis 3 had the most significant species correlations but accounted for the least variance of the 3 axes. Of the six species significantly correlated with axis 3 there are three native shrub species. However, the native low shrub species on this axis often have invasive tendencies and can be a major threat to floristic diversity in the area (Grant et al. 2020b). Similar to this study, Grant et al. (2020b) also found a significant correlation between these invasive shrubs and *Poa pratensis*, meaning it is likely to coexist with invasive shrubs. However, many other species positively correlated with the axis are desirable native species. Species negatively correlated with the axis are a mixture of invasive weedy species and desirable native species. Given the composition of species on both ends of the axis it appears to be controlled by invasive weedy versus shrub species. Unlike Grant et al. (2020b) it appears both are associated with native diversity rather than inhibiting it, but relative covers never reached 50-100% percent as it did with invasive grasses meaning they are not yet exhibiting as invasive of tendencies in the low prairie zone.

The major species driver for axis 1 of the wet meadow zones was *Phalaris arundinacea* (reed canarygrass). It had the strongest negative correlation with the axis (-0.919) with few other species being even slightly negatively correlated. It is evident in Figure 3.3 that sites on the positive end of axis 1 have low relative cover of *Phalaris arundinacea* while sites on the negative have very high relative covers. Most species were positively correlated with axis 1 and those with the highest positive correlations were a mixture of native and introduced species, meaning *Phalaris arundinacea* not only outcompetes native species, but other introduced species as well. Mullhouse and Galatowitsch et al. (2003) showed invasive perennials, particularly *Phalaris arundinacea*, are frequently occurring in restored wetlands with cover approaching 75-100%, often resulting in the absence of many common native wetland species. These results are similar to what was observed over the course of this study with *Phalaris arundinacea*, often nearing 100% relative cover in wet meadows on reseeded grassland. When those cover percentages were reached there was little diversity in those wet meadow zones. *Phalaris arundinacea* has many different mechanisms of invasion allowing it to control plant community composition. Its rapid growth, self-facilitation, ability to handle disturbance, and suppression of native seedlings are just a few ways it becomes dominant (Galtowitsch et al. 1999; Adams and Galatowitsch 2005).

The variance in species composition for axis 2 was likely the results of slightly brackish conditions. Almost all species with a significant or near significant negative correlation with axis 2 are considered primary species found in slightly brackish conditions for the wet meadow zone (Stewart and Kantrud 1971). The species negatively correlated with axis 2 are mostly a combination of freshwater wet meadow species and primarily upland species. Species significantly correlated both positively and negatively were also all more prevalent when there was low cover of *Phalaris arundinacea* but common species for wet meadows in reseeded grassland.

Both positive and negative correlations with axis 3 were associated primarily with native species. It is likely this axis shows species composition variance within relatively intact and native wetlands. The low number of intact or native temporary wetlands assessed is likely the reason it accounts for the least amount of variance of the 3 axes. There are far more species positively correlated than negatively correlated with axis 3. No species have a significant positive correlation with the axis, likely meaning none of those species were common or dominant enough to be considered a driving species. The two species significantly negatively correlated with the axis are native graminoids: *Calamagrostis canadensis* (bluejoint) and *Carex laeviconica* (smoothcone sedge). These were the only desirable native species with multiple instances of 50% or higher relative cover in the wet meadow zone of temporary wetlands. with those instances being when relative cover of *Phalaris arundinacea* was at 0%. These are both species associated with natural wetlands (Mullhouse and Galatowtisch 2003) and likely to be the dominant species in those wetlands when invasive species are absent or sparse.

There was not a significant difference ($p<0.05$) in the low prairie plant communities between grassland types (native versus reseeded) for temporary wetlands. This is likely because of low species richness observed in the low prairies for wetlands in native grassland which meant those low prairie areas were still often dominated by invasive *Bromus inermis* and *Poa pratensis* to a similar enough extent to wetlands in reseeded grasslands. You can see in Figures 3.1 and 3.2 that a high percentage of temporary wetlands in both native and reseeded grassland had high covers of one or both species.

There was a significant difference between native and reseeded grassland for wet meadow plant communities. Restored or reseeded wetlands often do not reflect the conditions or plant communities of their natural counterparts as restored wetlands are often lacking an established wet meadow area (Seabloom and van der Valk 2003a) or have a plant community dominated by invasive species (Mullhouse and Galatowitsch 2003; Seabloom and van der Valk 2003b; Myla et al. 2008; Smith et al.2016). Reseeded areas were often surrounded by agriculture land. Surface water runoff from agricultural land tends to increase the abundance of invasive species (Galatowitsch et al. 1999). *Phalaris arundinacea* in particular increases its competitive advantage with higher nitrogen levels from agricultural runoff (Green and Galatowitsch 2002). Looking at Figure 3.3, the dominant invasive species *Phalaris arundinacea* had high relative cover in many of the wetlands in reseeded grassland but only a few of those in native grassland. While some native species and environmental conditions (e.g. salinity) accounted for variance within the plant community dataset, the main contributor to overall plant community differences for native versus reseeded grassland seems to be the prevalence of *Phalaris arundinacea*.

3.3.3. Seasonal Wetlands

Non-metric multidimensional scaling (NMS) ordination produced 3 axes for the low prairie, wet meadow, and shallow marsh zones of seasonal wetlands. The amount of variation explained by each axes follows numerical order (1-3). Each axis is used to explain the variation among plant communities seen in the 141 seasonal wetlands assessed. Similar to temporary wetlands, the axis explaining the most variation for each zone appeared to be controlled by cover of the most dominant invasive species.

Axis 1 for the low prairie zone appears to be controlled by *Bromus inermis* (smooth brome) cover. Very few species were negatively correlated along the axis with *Bromus inermis* being the lone species with a significant negative correlation (-0.895). On the positive end of the axis was a mixture of native and invasive species. While only one species has a significant positive correlation with the axis, several other were near to be considered significantly correlated (≥0.40). This is again indicating smooth brome is outcompeting all other species, not just native species. It was observed as a primary species in the low prairie of 133 of 141 seasonal wetlands with relative cover often over 50%. Figure 3.4 shows the correlation of *Bromus inermis* to axis 1. Sites with high cover of *Bromus inermis* are nearer to the negative end and sites with low cover are nearer to the positive end. While other research has posed *Poa pratensis* (Kentucky bluegrass) as a major threat to plant community biodiversity (Murphy and Grant 2005; Grant et al. 2009; Grant et al. 2020b) it is evident here *Bromus inermis* is currently the greatest threat, at least in low prairie plant communities we surveyed, often even outcompeting *Poa pratensis*.

Axis 2 for the low prairie zone is likely controlled by native versus reseeded grassland. All species positively correlated with axis 2, apart from *Poa pratensis*, are native species. Many of the species negatively correlated with the axis are those used in FWS fee-title land reseeding efforts prior to their updated management goals (Dixon et al. 2019). *Poa pratensis* and low shrubs are again correlated similar to Grant et al. (2020b). Unlike observations of the true uplands, these species seem to coexist with native plant species in the low prairie. In fact, looking at Figure 3.5, *Poa pratensis* often had its highest relative cover at native prairie wetland sites. However, higher cover of low shrubs and *Poa pratensis* at these sites is more likely due to less competition from *Bromus inermis* rather than these species facilitating native diversity.

On axis 3 there is a trend showing competition between native species and *Poa pratensis* as would be expected based on previous research (Murphy and Grant 2005; Grant et al. 2009; Grant et al. 2020b). A majority of species positively correlated to axis 3 are native grasses and forbs, while a much higher percentage of species negatively correlated are introduced species, with *Poa pratensis* being the most prominent. While *Poa pratensis* rarely reached relative covers nearing 100% as *Bromus inermis* did, it did often reach covers of 25-75% percent which is enough to significantly reduce native species cover and possibly diversity similar to what was seen in Grant et al. (2020b).

Axis 1 of the wet meadow zone was primarily driven by *Phalaris arundinacea* (reed canarygrass) cover. Like previous axes dominated by one invasive species, it appears to be outcompeting all other species, invasive species included, to the point where its presence is the single largest driver for plant community composition in the zone. When present it often had a relative cover nearing 100% leaving few opportunities for any other species to colonize. These results concur with previous studies showing the negative effects it has on plant community diversity (Green and Galatowitsch 2002; Perry et al. 2004; Adams and Galatowitsch 2005; Schooler et al. 2006; Spyreas et al. 2010). Figure 3.6 shows the obvious trend for relative cover of *Phalaris arundinacea* driving axis 1.

Axis 2 appears to be following a species trend of freshwater to moderately brackish wet meadows of seasonal wetlands (Steward and Kantrud 1971). Most species with a significant or near significant correlation to the axis are native species because those species evolved with the natural variability in hydrology and salinity found in PPR wetlands (Richardson and Verpraskas 2001; Euliss et al. 2004). This variability is likely due to the relative position on the landscape indicating recharge or flow-through seasonal wetlands but may also be influenced by regional distribution. Most species with a significant or near significant positive correlation to the axis are primary or secondary species found in wet meadows of freshwater wetlands while all of those with a significant or near significant negative correlation are primary or secondary species found in moderately brackish wet meadows of seasonal wetlands (Steward and Kantrud 1971).

There was no obvious trend for species composition on axis 3. Both species significantly correlated with axis 3 were common species in the case of both high and low *Phalaris arundinacea* cover. It is likely because of this they are the next two most dominant species in reseeded wetlands. Both were frequently found as primary species and had moderate cover when *Phalaris arundinacea* was present. It is possible axis 3 shows the differences in species composition of reseeded wetlands regardless of *Phalaris arundinacea* cover.

The major driver for axis 1 of the shallow marsh zone was relative cover of *Typha x glauca* (hybrid cattail). When present, *Typha x glauca* cover often neared 100% cover. *Typha x glauca* has been known to form monocultures (Bansal et al. 2019) in wetlands and has been known to significantly lower wetland diversity (Larkin et al. 2012; Smith et al. 2016). It is often most prevalent in disturbed wetlands (Ralston et al. 2007). This can be seen in Figure 3.7, which shows a majority of native sites having low relative cover while a majority of reseeded sites have high relative cover of *Typha x glauca*. Looking at Figure 3.7, it is also evident *Typha x glauca* is the main driver of axis 1 as all sites with high cover appear on the negative end of the axis and all sites with low cover appear on the positive end.

Axis 2 appears to be controlled by wetland hydrology. Seasonal, annual, and interannual natural variability in hydrology is normal in PPR wetlands and ultimately changes the plant community composition (Euliss et al. 2004). Many species positively correlated with the axis are primary or secondary species in the natural drawdown phase (Steward and Kantrud 1971). Most species negatively correlated with the axis are primary or secondary species associated with the normal emergent phase of seasonal wetlands. The species composition appears not to be influenced by brackish or freshwater conditions. Due to the natural variability in hydrology this axis could be influenced by the time of year each assessment took place, seeing as seasonal wetlands often dry out and slightly change plant community composition later in the growing season (Euliss et al. 2004). It could also be influenced by study year. Year 2 of the study was significantly drier than year 1 which could change the plant community composition on a regional scale (i.e. higher occurrence of natural drawdown species).

Phalaris arundinacea is the most significantly correlated species with axis 3 (0.545). Seeing as *Phalaris arundinacea* was commonly a dominant shallow marsh species with relative cover often reaching 50-90% when Typha x glauca was sparse or absent, it is possible axis 3 is controlled by dominance of *Phalaris arundinacea*. It does not exhibit the same level influence on plant community composition as it does in the wet meadow zone, but still has the ability to dominate the shallow marsh of disturbed wetlands where *Typha x glauca* has not yet established itself because of its ability to withstand varying levels of hydrology and soil saturation (Kellogg et al. 2003).

There was a significant difference between native versus reseeded grassland for all zones of seasonal wetlands. It was clear from the graphical outputs of NMS ordination that reseeded wetland sites had much higher cover of the dominant invasive species in each zone. With those dominant invasive species accounting for so much of the variation in plant community composition for each zone it is logical to assume they are largely responsible for the difference in plant community composition between grassland types. This difference between reseeded and native wetlands has been commonly documented in studies looking at plant community composition (Mullhouse and Galatowitsch 2003, Seabloom and van der Valk 2003a; Seabloom and van der Valk 2003b; Myla et al. 2008; Smith et al.2016). Past disturbance may not be as large of a factor in seasonal wetlands as it is in temporary wetlands because the central shallow marsh zone was not as frequently cultivated as the wet meadow zone (Galatowitsch and van der Valk 1996b). However, any formerly cultivated wetlands have undergone major disturbance in the past which gives invasive species the completive advantage (Galatowitsch et al. 2000; Ralston et al. 2007). This allows these species to form monocultures (Bansal et al. 2019) and reduce native diversity, ultimately making them the major driving factors in current plant community composition as a result of past disturbance.

3.3.4. Conclusion

This study looked to further assess the plant community composition and the species composition drivers for wetlands on FWS fee-title lands in the PPR. Results echo previous studies showing plant community composition is significantly different between native and reseeded wetlands. The largest driver influencing plant community composition is abundance of dominant invasive species. Major driving species include *Bromus inermis*, *Poa pratensis*, *Phalaris arundinacea*, and *Typha x glauca*. These species dominate in each corresponding zone of most of the more highly disturbed wetlands (i.e. previously cultivated) and are often absent or sparse in more pristine native wetlands. They are often replacing diverse communities of native species with monocultures. Other introduced and native species show tendencies to control plant community composition but are most often outcompeted by the dominant invasive species.

Native grasslands in some areas appear to remain fairly intact and minimally influenced by dominant invasive species. However, former disturbance in reseeded wetlands is obvious and wetlands still remain degraded many years after restoration. Unless there are breakthroughs in invasive species control, the plant communities of reseeded wetlands are unlikely to resemble those of native wetlands in the near future. These results do not provide for much optimism for reseeded wetlands but still offer a foundation of plant community data to help guide FWS staff in attempts to alter, restore, or maintain native plant communities.

3.3.5. Future Research Needs

This study contributes to the understanding of wetland plant community composition and drivers of that composition but leaves many questions and possibilities for PPR wetland research in the future. Continual monitoring of sites is necessary to obtain information regarding plant community changes over time, especially in native prairie areas. PPR wetlands are dynamic ecosystems undergoing constant change due to natural and anthropogenic disturbance ultimately affecting plant community composition (Galatowitsch et al. 2000; Euliss et al. 2004).

Similar to NPAM, wetland sites will need to be revisited in coming years to determine how current management is affecting wetland plant communities. Invasive species were a major driver for plant community composition of wetlands just as they are for upland sites (Grant et al. 2009; Grant et al. 2020a; Grant et al. 2020b). Factors influencing the cover of these invasive species need to be further studied so they can be properly managed in hopes of decreasing invasive species cover on reseeded wetlands and maintaining native species diversity in native wetlands. This requires continual monitoring of established wetland sites for which there already is a record of the plant community composition. This study has provided a base with established sites to do continual monitoring and observe plant community changes in hopes a management program can be made to maintain or increase native wetland species diversity and ecological integrity.

With the data collected from this study there are many different variations of analysis to be conducted to further aid in understanding wetland plant community composition and drivers. This study analyzed data in a specific manner of grouping communities by wetland classification, then by vegetation zone, and deleting any species which did not appear within roughly 5% of wetland sites for that classification. Many species were only observed a few times due to being in different habitats than the most common species (i.e. wetlands in pristine native prairie or forested situations). This meant excluding a high percentage of primary species from each group. NMS ordination could be done on plant communities within more specific groups with similar habitat characteristics (i.e. forested wetlands, native prairie wetlands, or reseeded wetlands surrounded by agriculture) to determine the plant community drivers within more specific ecosystems rather than as a whole. This would likely result in less primary species needing to be excluded from the dataset and would allow for an opportunity to compare plant community drivers between specific ecosystems and the region as a whole. It would also simply show which species are occurring or missing in specific ecosystems or under certain disturbances similar to Galatowitsch and van der Valk (1996a) and Galatowitsch et al. (2000).

Grant et al. (2020b) looked at composition of both plant groups and individual species for upland plant communities while comparing analyzing them along patterns in precipitation and temperature in the region. The data collected from this study could be grouped similarly (e.g. low shrubs, native forbs, or weedy forbs) with some of the major driving invasive species left at the species level. This would provide an opportunity to examine differences in the way species interact in wetlands versus uplands. For example, Grant et al. (2020) found strong correlations between *Poa pratensis* (Kentucky bluegrass) and low shrub cover; both of which corresponded with decreased cover of native grasses and forbs. In our study, we found positive correlations between low shrub species, *Poa pratensis*, and many desirable native species. This could not only provide insight into why species are behaving differently in the true uplands versus the low prairie, but could also show which native species can coexist most easily with high cover of invasive species. Knowing this will be useful for maintaining some level of diversity at sites where invasive species cannot be controlled.

Lastly, because of the influence of invasive species on the overall plant communities and the difficulties these species are presenting to land managers, it is crucial to not only focus future research on best ways to manage these species, but also to recognize which other introduced species are likely to cause similar issues in the future. Our data shows certain introduced species that are often not thought of as highly invasive, such as *Phleum pratense* (timothy grass) or *Alopecurus arundinaceus* (creeping foxtail), are exhibiting invasive tendencies and have very high cover at some sites. Some of these species are still commonly used in seeing mixes and as forage for animals. Van Kluenen et al. (2020) provides a model showing the naturalization of introduced species in relation to their economic use. Plants introduced for animal feed have high rates of naturalization success. This model, along with cover data gathered, could be used to help predict which species are becoming highly naturalized and may cause major problems for species diversity and ecosystem integrity in the future.

3.3.6. Management Implications

Plant community composition is an important aspect to understand in order for the FWS to reach their management goals of increased diversity and ecological integrity as described by Dixon et al. (2019). Even with the results of this study and follow up future research there will still be many unanswered questions regarding how to best manage the land for those goals. This study, however, does provide the baseline needed to allow for future research that will help guide management decisions.

Results from this study made it clear invasive species are driving plant community composition of PPR wetlands and should be the focus of future wetland management. This may involve determining methods of reduction, control, or prevention of invasive species takeovers. As of now, it appears invasive species have taken over much of the previously cultivated FWS fee-title land, but some areas of native grassland remain intact with high native species diversity and cover. While all forms of invasive species control need to be considered, maintaining the native diversity and ecological integrity of our native prairie wetlands should be the highest priority, seeing as these relatively few areas are harboring such a high percentage of desired native species compared to the more common reseeded wetlands. With invasive species being sparse in many of the native prairie wetlands, prevention of increased abundance seems a more realistic goal than completely transforming the landscapes where they have already taken over.

Attempting to reflect natural disturbance of the past seems the most common and logical way to control the spread of invasive species. It has been proven to work to some extent in the uplands (Grant et al. 2009; Dixon et al. 2019) so similar management should work for wetlands as well since they evolved with the same natural disturbance. Grazing seems to be a widely available resource in many parts of the PPR, but fire as a management tactic is not as widely accepted or applied. In order for native prairie wetlands to maintain their species diversity and ecological integrity, grazing efforts need to be maintained and properly monitored, and prescribed burning needs to be more widely applied. These land management efforts also need to be coupled with an attempt to increase public understanding of invasive species and the issues they are causing in natural systems. Results of this study and future monitoring efforts can be used to inform land managers and the general public why controlling invasive species is so vital to maintaining the ecological integrity of PPR wetlands and all of the ecosystem services they provide.

LITERATURE CITED

Adams, R.A., and S.M. Galatowitsch. 2005. Phalaris arundinacea (reed canary grass): Rapid growth and growth pattern in conditions approximating newly restored wetlands. Ecoscience 12(4):569–573.

Adams, R.A., and S.M. Galatowitsch. 2006. Increasing the effectiveness of reed canary grass (Phalaris arundinacea L.) control in wet meadow restorations. Restoration ecology 14(3):441–451.

Anteau, M.J., A.D. Afton, A.C.E. Anteau Anteau, and E.B. Moser. 2011. Fish and land use influence Gammarus lacustris and Hyalella azteca (Amphipoda) densities in large wetlands across the upper Midwest. Hydrobiologia 664(1): 69–80.

Badiou, P., R. McDougal, D. Pennock, and B. Clark. 2011. Greenhouse gas emissions and carbon sequestration potential in restored wetlands of the Canadian Prairie Pothole Region. Wetlands Ecology and Management 19(3):237–256.

Bansal, S., S.C. Lishawa, S. Newman, B.A. Tangen, D. Wilcox, D. Albert, M.J. Anteau, M.J. Chimney, R.L. Cressey, E. DeKeyser, K.J. Elgersma, S.A. Finkelstein, J. Freeland, R. Grosshans, P.E. Klug, D.J. Larkin, B.A. Lawrence, G. Linz, J. Mayburger, G. Noe, C. Otto, N. Reo, J. Richards, C. Richardson, L. Rodgers, A.J. Schrank, D. Svedarsky, S. Travis, N. Tuchman, and L. Windham-Myers. 2019. Typha (Cattail) invasion in North American wetlands. Biology, regional problems, impacts, ecosystem services, and management. Wetlands 39:645-684.

Balas, C.J., N.H., Jr., Euliss, and D.M. Mushet. 2012. Influence of conservation programs on amphibians using seasonal wetlands in the Prairie Pothole Region. Wetlands 32(2):333–345.

Ball, J.P. 1990. Influence of subsequent flooding depth on Typha control by burning and mowing. Journal of Aquatic Plant Management 28:32-36.

Berry, P., F. Yassin, R. Grosshans, and K.E. Lindenschmidt. 2017. Surface water retention systems for cattail production as a biofuel. Journal of Environmental Management 203:500-509.

Bluemle, J.P. 2000. The face of North Dakota—The geologic story (3d ed.): Bismarck, N. Dak., North Dakota Geological Survey Educational Series 26: 210 p.

Bourdaghs, M., C.A. Johnston, and R.R. Regal. 2006. Properties and performance of the Floristic Quality Index in Great Lakes coastal wetlands. Wetlands 26(3):718-735.

Bried, J.T., S.K. Jog, and J.W. Matthews. 2013. Floristic quality Assessment signals human disturbance over natural variability in a wetland system. Ecological Indicators 34:260-267.

Coops, H., and G. Vandervelde. 1995. Seed dispersal, germination and seedling growth of six helophyte species in relation to water-level zonation. Freshwater Biology 34:13–20.

Cowardin, L.M, V. Carter, F.C. Golet, and E.T. LaRoe. 1979. Classification of wetlands and deepwater habitats of the United States. U.S. Fish and Wildlife Service. FWS/OBS-79/31, Washington, DC.

Dahl, T.E. 2014. Status and trends of prairie wetlands in the United States 1997 to 2009: Washington, D.C., U.S. Fish and Wildlife Service: 67 p.

DeBerry, D.A., and J.E. Perry. 2015. Using the floristic quality concept to assess created and natural wetlands: Ecological and management implications. Ecological Indicators 53:247-257.

DeKeyser, E.S. Professor. Natural Resource Management, North Dakota State University. Fargo, N.D.

DeKeyser, E.S., D.R. Kirby, and M.J. Ell. 2003. An index of plant community integrity—Development of the methodology for assessing prairie wetland plant communities. Ecological Indicators 3(2):119–133.

DeKeyser, E.S., M. Biondini, D. Kirby, and C. Hargiss. 2009. Low prairie plant communities of wetlands as a function of disturbance: Physical parameters. Ecological Indicators 9:296-306.

Dillemuth, F.P., E.A. Rietschier, and J.T. Cronin. 2009. Patch Dynamics of a native grass in relation to the spread of invasive smooth brome (Bromus inermis). Biological Invasions 11(6): 1381-1391.

Dixon, C., S. Vacek, and T. Grant. 2019. Evolving management paradigms on U.S. Fish and Wildlife Service lands in the Prairie Pothole Region. Rangelands 41(1):36–43.

Duebbert, H.F., and H.A. Kantrud. 1974. Upland duck nesting related to land use and predator reduction. Journal of Wildlife Management 38(2):257-265.

Duebbert, H.F., E.T. Jacobson, K.F. Higgins, and E.B. Podoll. 1981. Establishment of seeded grasslands for wildlife habitat in the Prairie Pothole Region. Jamestown, ND, USA. U.S. Fish and Wildlife Service. Special Scientific Report 234: 21p.

Dupler, K., M. Guidugli-Cook, D.R. Brown, and S.C. Richter. 2020. Rapid assessment of wetland condition reflects amphibian community composition. Wetlands 40:451-464.

Elgersma, K.J., J.P. Martina, D.E. Goldberg, and W.S. Currie. 2017. Effectiveness of cattail (Typha spp.) management techniques depends on exogenous nitrogen inputs. Elemnta: Science of Anthropocene 5.

Ervin, G.N., B.D. Herman, J.T. Bried, and D.C. Holly. 2006. Evaluating non-native species and wetland indicator status as components of wetlands floristic assessment. Wetlands 26(4):1114-1129.

Eswaran, H., E. van der Berg, and P. Reich. 1993. Organic carbon in soils of the world. Soil Science Society of America Journal 57:192–194.

Euliss, N.H., and D.M. Mushet. 1996. Water-level fluctuation in wetlands as a function of landscape condition in the Prairie Pothole Region. Wetlands 16(4):587–593.

Euliss, N.H., and D.M. Mushet. 2011. A multi-year comparison of IPCI scores for Prairie Pothole wetlands: Implications of temporal and spatial variation. Wetlands 31:713-723.

Euliss, N.H., J.W. LaBaugh, L.H. Fredrickson, D.M. Mushet, M.K. Laubhan, G.A. Swanson, T.C. Winter, D.O. Rosenberry, and R.D. Nelson. 2004. The wetland continuum: a conceptual framework for interpreting biological studies. Wetlands 24(2):448-458.

Euliss, N.H., R.A. Gleason, A. Olness, R.L. McDougal, H.R. Murkin, R.D. Robarts, R.A. Bourbonniere, and B.G. Warner. 2006. North American prairie wetlands are important nonforested land-based carbon storage sites. Science and the Total Environment 361:179-188.

Fennessy, M.S., M. Gernes, J. Mack, and D.H. Wadrop. 2002. Methods for evaluating wetland condition: using vegetation to assess environmental conditions in wetlands. EPA-822-R-02-020. U.S. Environmental Protection Agency. Office of Water: Washington, DC.

Fennessy, M.S, A.D. Jacobs, and M.E. Kentula. 2007. An evaluation of Rapid Assessment Methods for assessing the ecological condition of wetlands. Wetlands 27(3):543-560.

Galatowitsch, S.M., and A.G. van der Valk. 1996a. The vegetation of restored and natural prairie wetlands. Ecological Applications 6(1):102-112.

Galatowitsch, S.M., and A.G. van der Valk. 1996b. Vegetation and environmental conditions in recently restored wetland in the Prairie Pothole Region of the USA. Vegetation 126(1):89-99.

Galatowitsch, S.M., N. O. Anderson, and P. D. Ascher. 1999. Invasiveness in wetland plants in temperate North America. Wetlands 19(4):733–755.

Galatowitsch, S.M., D.C. Whited, R. Lehtinen, J. Husveth, and K. Schik. 2000. The vegetation of wet meadows in relation to their land-use. Environmental Monitoring and Assessment 60:121-144.

Gallaway, S., C. Davis, D. Dvorett, and B. Tramell. 2019. Evaluating the effectiveness of Floristic Quality Assessment as a tool for determining condition of depressional wetlands across ecoregions. Ecological Indicators 102:488-496.

Gallaway, S., C. Davis, D. Dvorett, and B. Tramell. 2020. Validation of the Oklahoma Rapid Assessment Method (OKRAM) in depressional wetlands using EPA's three-tiered framework. Wetlands 40:925-937.

Geddes, P., L. Murhpy, Y. Astudillo-Scalia, D. Blasini, S. Nugent, M.J. Rios, A.E. Schirmer. And J.P. Olfelt. 2021. Microsatellite markers reveal unprecedented high frequencies of hybridization among Typha species in the Midwestern US. Wetlands 41(2): http://dx.doi.org/10.1007/s13157-021-01429-2.

Gianopulos, K. 2018. Performance of rapid floristic indices for increasing cost-effectiveness of wetland condition evaluation. Ecological Indicators 95:502-508.

Gilbert, M.C., P.M. Whited, E.J. Clairain, and R.D. Smith. 2006. A regional guidebook for applying the hydrogeomorphic approach to assessing wetland functions of Prairie

Potholes. ERDC/EL TR-06-5. United States Army Engineer Research and Development Center: Vicksburg.

Gleason, R.A., and N.H. Euliss Jr. 1998. Sedimentation of prairie wetlands. Great Plains Research 8:97–112.

Gleason, R.A., and B.A. Tangen. 2014. Brine contamination to aquatic resources from oil and gas development in the Williston Basin, United States. U.S. Geological Survey Scientific Investigations Report: 127 p.

Gleason, R.A., N.H. Euliss Jr., D.E. Hubbard, and W.G. Duffy. 2003. Effects of sediment load on emergence of aquatic invertebrates and plants from wetland soil egg and seed banks. Wetlands 23(1):26–34.

Gleason, R.A., N.H. Euliss Jr., B.A. Tangen, M.K. Laubhan, and B.A. Browne. 2011. USDA conservation program and practice effects on wetland ecosystem services in the Prairie Pothole Region. Ecological Applications 21(3):65–81.

Grant, T.A., B. Flanders-Wanner, T.L. Shaffer, R.K. Murphy, and G.A. Knutsen. 2009. An emerging crisis across northern prairie refuges: Prevalence of invasive plants and a plan for adaptive management. Ecological Restoration 27:58-65.

Grant, T.A., T.L. Shaffer, and B. Flanders. 2020a. Resiliency of native prairies to invasion by Kentucky bluegrass, smooth brome, and woody vegetation. Rangeland Ecology & Management 73(2):321-328.

Grant, T.A., T.L. Schaffer, and B. Flanders. 2020b. Patterns of smooth brome, Kentucky bluegrass, and shrub invasion in the Northern Great Plains vary with temperature and precipitation. Natural Areas Journal 40(1):11-22.

Green, E.K., and S.M. Galatowitsch. 2002. Effects of Phalaris arundinacea and Nitrate-N addition on the establishment of wetland plant communities. Journal of Applied Ecology 39(1):134-144.

Greer, A.K., B.D. Dugger, D.A. Graber, and M.J. Petrie. 2007. The effects of seasonal flooding on seed availability for spring migrating waterfowl. The Journal of Wildlife Management 71:1561-1566.

Groffman, P.M., J. Baron, T. Blett, A. Gold, I. Goodman, L. Gunderson, B. Levinson, M. Palmer, H. Paerl, G. Peterson, N. Poff, D. Rejeski, J. Reynolds, M. Turner, K. Weathers, and J. Wiens. 2006. Ecological thresholds: the key to successful environmental management or an important concept with no practical application? Ecosystems 9(1):1-13.

Hager, H.A. 2004. Competitive effect versus competitive response of invasive and native wetland plant species. Oecologia 139(1):140-149.

Hanson, M.A., K.D. Zimmer, M.G. Butler, B.A. Tangen, B.R. Herwig, and N.H. Euliss Jr. 2005. Biotic interactions as determinants of ecosystem structure in prairie wetlands—An example using fish. Wetlands 25(3):764–775.

Hargiss, C.L.M., E.S. DeKeyser, D.R. Kirby, and M.J. Ell. 2008. Regional assessment of wetland plant communities using the index of plant community integrity. Ecological Indicators8(3):303–307.

Hargiss, C.L.M. 2009. Estimating wetland quality for the Missouri Coteau ecoregion in North Dakota. Fargo, North Dakota State University, Ph.D. dissertation: 162 p.

Hargiss, C.L.M., E.S. DeKeyser, J.E. Norland, and M.J. Ell. 2017. Comparing tiers of a multi-tiered wetland assessment in the Prairie Pothole Region. Wetland Ecology and Management 25(5):639–647.

Hayashi, M., G. van der Kamp, and D.O. Rosenberry. 2016. Hydrology of prairie wetlands: Understanding the integrated surface-water and groundwater processes. Wetlands 36(2):237-254.

Hegazy, A.K., N.T. Abdel-Ghani, and G.A. El-Chaghaby. 2011. Phytoremediation of industrial wastewater potentiality by Typha domingensis. International journal of Environmental Science and Technology 8:639-648.

Herlihy, A.T., J. Sifneos, C. Bason, A. Jacobs, M.E. Kentula, and M.S. Fennessy. 2009. An approach for evaluating the repeatability of Rapid Assessment Methods: The effects of training and experience. Environmental Management 44:369-377.

Herlihy, A.T., M.E. Kentula, T.K. Magee, G.A. Lomnicky, A.M. Nahlik, and G. Serenbetz. 2019. Striving for consistency in the National Wetland Condition Assessment: developing a reference condition approach for assessing wetlands at a continental scale. Environmental Monitoring and Assessment 191(1):327-346.

Herwig, B.R., K.D. Zimmer, M.A. Hanson, M.L. Konsti, J.A. Younk, R.W. Wright, S.R. Vaughn, and M.D. Haustein. 2010. Factors influencing fish distributions in shallow lakes in prairie and prairie-parkland regions of Minnesota, USA. Wetlands 30(3):609–619.

Hossack, B.R., K.L. Smalling, C.W. Anderson, T.M. Preston, I.M. Cozzarelli, and R. Ken Honeycutt. 2018. Effects of persistent energy-related brine contamination on amphibian abundance in national wildlife refuge wetlands. Biological Conservation 228:36–43.

Johnson, R.R., F.T. Oslund, and D.R. Hertel. 2008. The past, present, and future of prairie potholes in the United States. Journal of Soil and Water Conservation 63(3):84–87.

Johnson, W.C. 2019. Ecosystem services provided by prairie wetlands in northern rangelands. Rangelands 41(1):44-48.

Johnson, W.C., B.V. Millett, G. Tagir, R.A. Voldseth, G.R. Guntenspergen, and D.E. Naugle. 2005. Vulnerability of northern prairie wetlands to climate change. Bioscience. 55 (10):863-872.

Johnson, W.C., B. Werner, G.R. Guntenspergen, R.A. Voldseth, B. Millett, D.E. Naugle, M.R. Tulbure, W.H. Carroll, J. Tracy, and C. Olawsky. 2010. Prairie wetland complexes as landscape functional units in a changing climate. Bioscience 60(2):128-140.

Johnson, W.C., B. Werner, and G.R. Guntenspergen. 2016. Non-linear responses of glaciated prairie wetlands to climate warming. Climatic change 134(1-2):209-223.

Johnston, C.A. 2013. Wetland losses due to row crop expansion in the Dakota Prairie Pothole Region. Wetlands 33:175-182.

Johnston, C.A., and N.E. McIntyre. 2019. Effects of cropland encroachment on Prairie Pothole wetlands: numbers, density, size, shape, and structural connectivity. Landscape Ecology 34:827-841.

Jordan, N.R., D.L Larson, and S.C. Huerd. 2008. Soil modification by invasive plants: effects on native and invasive species of mixed-grass prairies. Biological Invasions 10(2):177-190.

Jung, J.A., H.N. Rogers, and G.P. Grabas. 2020. Refinement of an index of ecological condition for marsh bird communities in lower Great Lakes coastal wetlands. Ecological Indicators 113: https://doi.org/10.1016/j.ecolind.2020.106097.

Kantrud, H.A., G.L. Krapu, and G.A. Swanson. 1989. Prairie basin wetlands of the Dakotas: a community profile. U.S. Fish and Wildlife Service. Washington DC, USA. Biological Report 85 (7.28).

Karr, J.R. 1981. Assessment of biotic integrity using fish communities. Fisheries 6(6):21–27.

Karr, J.R., and D.R. Dudley. 1981. Ecological perspective on water quality goals. Environmental Management 5:55–68.

Katterer, T., and O. Andren. 1999. Growth dynamics of reed canarygrass (Phalaris arundinacea L.) and its allocation of biomass and nitrogen below ground in a field receiving daily irrigation and fertilisation. Nutrient Cycling in Agroecosystems 54(1):21–29.

Kellogg, C.H., S.D. Bridgham, and S.A. Leicht. 2003. Effects of water level, shade and time on germination and growth of freshwater marsh plants along a simulated successional gradient. Journal of Ecology 91:274-282.

Kentula, M.E, and S.G. Paulsen. 2019. The 2011 national wetland condition assessment: overview and an invitation. Environmental Monitoring and Assessment 19(1):325-342.

Kephart, K.D., and D.R. Buxton. 1993. Forage quality responses of C3 and C4 perennial grasses to shade. Crop Science 33(4):831–837.

Kincaid, T., and T. Olsen. 2019. spsurvey: spatial survey design and analysis. R package version 4.0.0: 195 p.

Kirsch, E.M., B.R. Gray, T.J. Fox, and W.E. Thogmartin. 2007. Breeding bird territory placement in riparian wet meadows in relation to invasive reed canary grass, Phalaris arundinacea. Wetlands 27(3):644–655.

Knutsen, G.A., and N.H. Euliss Jr. 2001. Wetland restoration in the Prairie Pothole Region of North America: A literature review. USGS/BRD/BSR-2001-006 Biological Science Report, US Geological Survey, Biological Resources Division, Reston, VA.

Krol, D.J., M.B. Jones, M. Williams, O. Ni Choncubhair, and G.J. Lanigan. 2019. The effect of land use change from grassland to bioenergy crops Miscanthus and reed canary grass on nitrous oxide emissions. Biomass & Bioenergy 120:396–403.

Kuthcer, T.E., and G.E. Forrester. 2018. Evaluating how variants of the Floristic Quality Assessment indicate wetland condition. Journal of Environmental Management 217:231-239.

Larkin, D., M. Freyman, S. Lishawa, P. Geddes, and N. Tuchman. 2012. Mechanisms of dominance by the invasive hybrid cattail Typha × glauca. Biological Invasions 14(1):65-77.

Larson, D.L., P.J. Anderson, and W. Newton. 2001. Alien plant invasion in mixed-grass prairie: effects of vegetation type and anthropogenic disturbance. Ecological Applications 11(1):128-141.

Larson, D.L., R.A. Royer, and M.R. Royer. 2006. Insect visitation and pollen deposition in an invaded prairie plant community. Biological Conservation 130:148-159.

Larson, D.L. D.L. Hernandez, J.L. Larson, J.B. Leone, and N. Pennarola. 2020. Management of remnant tallgrass prairie by grazing or fire: effects on plant communities and soil properties. Ecosphere 11(8): https://doi.org/10.1002/ecs2.3213.

Lavergne, S., and J. Molofsky. 2004. Reed Canary Grass (Phalaris arundinacea) as a biological model in the study of plant invasions. Critical reviews in plant sciences 23(5):415–429.

Lissey, A. 1971. Depression-focused transient groundwater flow patterns in Manitoba. Geological Association of Canada Special Paper 9:333-341.

Lopez, R.D., and M.S. Fennessy. 2002. Testing the Floristic Quality Assessment Index as an indicator of wetland condition. Ecological Applications 12(2):487-497.

Mack, J.J. 2007. Developing a wetland IBI with statewide application after multiple testing iterations. Ecological Indicators 7:864-881.

Marlor, K.M., C.R. Webster, and R.A. Chimner. 2014. Disturbance and wetland type alter reed canarygrass cover in northern Michigan. Invasive Plant Science and Management 7(1):121-131.

Matthews, J.W., P.A. Tessene, S.M. Wiesbrook, and B.W. Zercher. 2005. Effect of area and isolation on species richness and floristic quality in Illinois, USA wetlands. Wetlands 25(3):607-615.

Matthews, J.W., G. Spyreas, and C.M. long. 2015. A null model test of Floristic Quality Assessment: Are plant species' Coefficients of Conservatism Valid? Ecological Indicators 52:1-7.

McKenna, O.P., S.R. Kucia, D.M. Mushet, M.J. Anteau, and M.T. Wiltermuth. 2019. Synergistic interaction of climate and land-use drivers alter the function of North American, Prairie-Pothole wetlands. Sustainability 11(23):1-20.

McCune, B., and M.J. Mefford. 1999. PC–ORD—Multivariate analysis of ecological data, version 4. Gleneden Beach, Oreg., MjM Software Design.

McCune, B., J.B. Grace, and D.L. Urban. 2002. Analysis of ecological communities. Gleneden Beach, Oreg., MjM Software Design.

Millett, B., W.C. Johnson, and G. Guntenspergen. 2009. Climate trends of the North American Prairie Pothole Region 1906-2000. Climatic Change 93:243-267.

Mita, D., E. DeKeyser, D. Kirby, and G. Easson. 2007. Developing a wetland landscape condition prediction model using landscape structure variability. Wetlands 27(4):1124-1133.

Morton, J.F. 1975. Cattails (Typha spp.) – weed problem or potential crop? Economic Botany 29:7-29.

Mullhouse, J.M., and S.M. Galatowitsch. 2003. Revegetation of prairie pothole wetlands in the mid-continental US: twelve years post-reflooding. Plant Ecology 169:143-159.

Murphy, R.K., and T.A. Grant. 2005. Land management history and floristics in mixed-grass prairie, North Dakota, USA. Natural Areas Journal 25:351-358.

Mushet, D.M., N.H. Euliss, and T.L. Shaffer. 2002. Floristic Quality Assessment of one natural and three restored wetland complexes in North Dakota, USA. Wetlands 22(1):126-138.

Myla, F., J. Aronson, and S. Galatowitsch. 2008. Long-term vegetation development of restored Prairie Pothole wetlands. Wetlands 28(4):883-895.

Niemuth, N.D., B. Wangler, and R.E. Reynolds. 2010. Spatial and temporal variation in wet area of wetlands in the Prairie Pothole Region of North Dakota and South Dakota. Wetlands 30(6):1053–1064.

Ostrem, L. 1988a. Studies on genetic variation in reed canary grass, Phalaris arundinacea L. II. Forage yield and quality. Hereditas 108(2):103–113.

Ostrem, L. 1988b. Studies on genetic variation in reed canary grass, Phalaris arundinacea L. III. Seed yield and seed yield components. Hereditas 108(2):159–168.

Otte, M.L., W. Fang, and M. Jiang. 2021. A framework for identifying reference wetland conditions in highly altered landscapes. Wetland Restoration 41(40): https://doi.org/10.1007/s13157-021-01439-0.

Pennock, D., T. Yates, A. Bedard-Haughn, K. Phipps, R. Farrell, and R. McDougal. 2010. Landscape controls on N2O and CH4 emissions from freshwater mineral soil wetlands of the Canadian Prairie Pothole Region. Geoderma 155(3–4):308–319.

Perry, L.G., S.M. Galatowitsch, and C.J. Rosen. 2004. Competitive control of invasive vegetation: a native wetland sedge suppresses Phalaris arundinacea in carbon-enriched soil. Journal of Applied Ecology 41(1):151-162.

Poiani, K.P., W.C. Johnson, and T. Kittel. 1995. Sensitivity of a prairie wetland to increased temperature and seasonal precipitation changes. Water Resources Bulletin 31:283-294.

Preston, T.M., M.J. Borgreen, and A.M. Ray. 2018. Effects of brine contamination from energy development on wetland macroinvertebrate community structure in the Prairie Pothole Region. Environmental Pollution 239:722–732.

Ralston, S.T., G.M. Linz, W.J. Bleier, and H.J. Homan. 2007. Cattail distribution and abundance in North Dakota. Journal of Aquatic Plant Management 45:21-24.

Rashford, B.S., R.M. Adams, J. Wu, R.A. Voldseth, G.R. Gutenspergen, B. Werner, and W.C. Johnson. 2016. Impacts of climate change on land-use and wetland productivity in the Prairie Pothole Region of North America. Regional Environmental Change 16:515-526.

Reuter, D.D. 1986. Sedge meadows of the upper Midwest: a stewardship summary. Natural Areas Journal 6:27–34.

Richardson, J.L., and M.J. Verpraskas. 2001. Wetland soils genesis, hydrology, landscapes, and classification. Lewis Publishers. Boca Raton, FL. 417p.

Rooney, R.C., S.E. Bayley, I.F. Creed, and M.J. Wilson. 2012. The accuracy of land cover-based wetland assessments is influenced by landscape extent. Landscape Ecology 27:1321-1335.

Rothrock, E.P. 1943. The geology of South Dakota, part I—The surface. Vermillion, South Dakota Geological Survey Bulletin number 13:88 p.

Rothrock, P.E., T.P. Simon, and P.M. Stewart. 2008. Development, calibration, and validation of a littoral zone plant index of biotic integrity (PIBI) for lacustrine wetlands. Ecological Indicators 8(1):79-88.

Rubio-Cisneros, N., O. Aburto-Oropeza, J. Murray, C. Gonzalez-Abraham, J. Jackson, and E. Ezcurra. 2014. Transnational ecosystem services: the potential of habitat conservation for waterfowl through recreational hunting activities. Human Dimensions of Wildlife 19(1):1-16.

Samson, F., and F. Knopf. 1994. Prairie conservation in North America. Bioscience 44:418-421.

Schooler, S.S., P.B. McEvoy, and E.M. Coombs. 2006. Negative per capita effects of purple loosestrife and reed canary grass on plant diversity in wetland communities. Diversity and Distributions 12(4):351-363.

Seabloom, E.W., and A.G. van der Valk. 2003a. The development of vegetative zonation patterns in restored Prairie Pothole wetlands. Journal of Applied Ecology 40(1):92-100.

Seabloom, E.W., and A.G. van der Valk. 2003b. Plant diversity, composition, and invasion of restored and natural prairie pothole wetlands: implications for restoration. Wetlands 23(1):1-12.

Shaw, D.A., G. van der Kamp, F.M. Conly, A. Pietroniro, and L. Martz. 2012. The fil-and-spill hydrology of prairie wetland complexes during drought and deluge. Hydrological Processes 26:3147-3156.

Sifneos, J.C., A.T. Herlihy, D. Jacobs, and M.E. Kentula. 2010. Calibration of the Delaware Rapid Assessment Protocol to a comprehensive measure of wetland condition. Wetlands 30:1011-1022.

Smalling, K.L., C.W. Anderson, R.K. Honeycutt, I.M. Cozzarelli, T. Preston, and B.R. Hossack. 2019. Associations between environmental pollutants and larval amphibians in wetlands contaminated by energy-related brines are potentially mediated by feeding traits. Environmental Pollution 248:260–268.

Smart, M.D., R.S. Cornman, D.D. Iwanowicz, M. McDermott-Kubeczko, J.S. Pettis, M.S. Spivak, and C.R.V. Otto. 2017. A comparison of honey bee-collected pollen from working agricultural lands using light microscopy and ITS metabarcoding. Environmental Entomology 46(1):38–49.

Smith, C.L. 2011. Effects of sediment removal on vegetation communities in Prairie Pothole wetlands in North Dakota. Fargo, North Dakota State University, M.S. Thesis: 84p.

Smith, C., E.S. DeKeyser, C. Dixon, B. Kobelia, and A. Little. 2016. Effects of sediment removal on prairie pothole wetland plant communities in North Dakota. Natural Areas Journal 36:48-58.

Smith, K.L. 2020. Returning the northern mixed grass prairie to Lostwood. U.S. Fish & Wildlife Service Region 3 General Technical Report. 304p.

Smith, L.M., and J.A. Kadlec. 1985. Comparisons of prescribed burning and cutting of Utah marsh plants. The Great Basin Naturalist 45:462-466.

Spyreas, G., B.W. Wilm, A.E. Plocher, D.M. Ketzner, J.W. Matthews, J.L. Ellis, and E.J. Heske. 2010. Biological consequences of invasion by reed canary grass (Phalaris arundinacea). Biological invasions 12(5):1253–1267.

Standen, K.M., P.A. Chambers, and J.M. Culp. 2018. Arrowhead (Sagittaria cuneata) as a bioindicator of nitrogen and phosphorus for prairie streams and wetlands. Wetlands Ecology and Management 26:331-343.

Steen, V.A., S.K. Skagen, and B.R. Noon. 2014. Vulnerability of breeding waterbirds to climate change in the Prairie Pothole Region, U.S.A. PLoS ONE 9(6):1-14.

Steen, V.A., S.K. Skagen, and C.P. Melcher. 2016. Implications of climate change for wetland-dependent birds in the Prairie Pothole Region. Wetlands 36(2):445–459.

Stein, E.D., A.E. Fetscher, R.P. Clark, A. Wiskind, J.L. Grenier, M. Sutula, J.N. Collins, and C. Grosso. 2009. Validation of a wetland Rapid Assessment Method: use of EPA's level 1-2-3 framework for method testing and refinement. Wetlands 29(2):648–665.

Stevens Jr., D.L., and A.R. Olsen. 2004. Spatially balanced sampling of natural resources. Journal of the American Statistical Association 99(465):262–278.

Stevens Jr., D.L., and S.F. Jensen. 2007. Sample design, execution, and analysis for wetland assessment. Wetlands 27(3):515–523.

Stewart, R.E., and H.A. Kantrud. 1971. Classification of natural ponds and lakes in the glaciated prairie region. Washington, D.C., U.S. Fish and Wildlife Service, Resource Publication 92: 57 p.

Stewart, R.E., and H.A., Kantrud. 1972. Vegetation of Prairie Potholes, North Dakota, in relation to quality of water and other environmental factors. U.S. Geological Survey. Professional paper 585-D:36 p.

Stoddard, J.L., D.P. Larsen, C.P. Hawkins, R.K. Johnson, and R.H. Norris. 2006. Setting expectations for the ecological condition of streams: the concept of reference condition. Ecological Applications 16:1267-1276.

Swink, F.A., and G.S. Willhelm. 1994. Plants of the Chicago region. 4th ed. Indiana Academy of Science, Indianapolis.

Taft, J.B., G.S. Willhelm, D.M. Ladd, and L.A. Masters. 1997. Floristic Quality Assessment for vegetation in Illinois, a method for assessing vegetation integrity. Erigenia 15:3-95.

Tangen, B.A., M.G. Butler, and M.J. Ell. 2003. Weak correspondence between macroinvertebrate assemblages and land use in Prairie Pothole Region wetlands, USA. Wetlands 23(1):104–115.

Tangen, B.A., S. Bansel, R.R. Fern, E.S. DeKeyser, C.L.M. Hargiss, D.M. Mushet, and C.S. Dixon. 2019. Study design and methods for a wetland condition assessment on U.S. Fish and Wildlife Service fee-title lands in the Prairie Pothole Region of North Dakota, South Dakota, and Montana, USA. U.S. Fish and Wildlife Service. Open file report. 34p.

The Northern Prairie Great Plains Floristic Quality Assessment Panel (TNGPFQAP). 2001. Coefficients of conservatism for the vascular flora of the Dakotas and adjacent grasslands: U.S. Geological Survey, Biological Resources Division, Information and Technology Report USGS/BRD/ITR-2001-001. 32 p.

U.S. Army Corps of Engineers. 1987. Corps of Engineers Wetlands Deliniation Manual. Technical Report Y-87-1. U.S. Army Corps of Engineers Waterways Experiment Station. Vicksburg. MS. 100 p.

USDA, NRCS. 2021. National Plants Data Team. Greensboro, NC USA. http://plants.usda.gov.

Vaccaro, L.E., B.L. Bedford, and C.A. Johnson. 2009. Litter accumulations promotes dominance of invasive species of cattails (Typha spp.) in Lake Ontario wetlands. Wetlands 29(3):1036-1048.

Vamell, C.J., S.A. Thawaba, and M. Solis. 2010. Typha domingenius – a potential tool for bioremediation of wetlands as relevant to environmental forensics: a case study from Palo Verde, Costa Rica. Environmental Forensics 11:102-107.

van Kleunen, M., X. Xu, Q. Yang, N. Maurel, Z. Zhang, W. Dawson, F. Essl, H. Kreft, J. Pergl, P. Pyšek, P. Weigelt, D. Moser, B. Lenzner, and T.S. Fristoe. 2020. Economic use of plants is key to their naturalization success. Nature Communications 11(1):3201–3201.

van der Kamp, G., M. Hayashi, A. Bedard-Haughn, and D. Pennock. 2016. Prairie Pothole wetlands – suggestions for practical and objective definitions and terminology. Wetlands 36(2):229-235.

Werner, B.A., W.C. Johnson, and G.R. Guntenspergen. 2013. Evidence for 20th century climate warming and wetland drying in the North American Prairie Pothole Region. Ecology and Evolution 3(10):3471–3482.

Whittaker, R.H., and G.E. Likens. 1973. Primary production: the Biosphere and man. Human Ecology 1:357-369.

Wigand, C., B. Carlisle, J. Smith, M. Carullo, D. Fillis, M. Charpentier, R. McKinney, R. Johnson, and J. Heltshe. 2011. Development and validation of Rapid Assessment Indices of condition for coastal tidal wetlands in southern New England, USA. Environmental Monitoring and Assessment 182:31–46.

Willhelm, G.S., and D.M. Ladd. 1988. Natural area assessment in the Chicago region. Transactions of the 53rd North American Wildlife & Natural Resources Conference. Pp. 361-375.

Wilson, M.J., and S.E. Bayley. 2012. Use of single versus multiple biotic communities as indicators of biological integrity in northern prairie wetlands. Ecological Indicators 20:187-195.

Wilson, M.J., A.S. Forrest, and S.E. Bayley. 2013. Floristic Quality Assessment for marshes in Alberta's northern prairie and boreal regions. Aquatic Ecosystem Health & Management. 16(3):288-299

Winter, T.C. 2000. The vulnerability of wetlands to climate change: a hydrologic landscape perspective. Journal of the American Water Resources Association 36(2):305-311.

Winter, T.C., and D.O. Rosenberry. 1995. The interaction of ground water with prairie pothole wetlands in the Cottonwood Lake area, east-central North Dakota, 1979–1990. Wetlands 15(3):193–211.

Zheng, T., N.B. Sutton, P. de Jager, R. Grosshans, S. Munira, and A. Farenhorst. 2017. Glyphosate (ab)sorption by shoots and rhizomes of native versus hybrid cattail (Typha). Bulletin of Environmental Contamination and Toxicology 99:595-600.

Zimmer, K.D., M.A. Hanson, and M.G. Butler. 2000. Factors influencing invertebrate communities in prairie wetlands—A multivariate approach. Canadian Journal of Fisheries and Aquatic Sciences 57(1):76–85.

Zimmer, K.D., M.A. Hanson, and M.G. Butler. 2002. Effects of fathead minnows and restoration on prairie wetland ecosystems. Freshwater Biology 47(11):2071–2086.

APPENDIX A. PRIMARY AND SECONDARY PLANT SPECIES ENCOUNTERED DURING IPCI ASSESSMENTS

Scientific Name[a]	Common Name[b]	C[c]	Life[d]
Acer negundo	Box Elder	1	P
Achillea millefolium	Common Yarrow	3	P
Agoseris glauca	Pale Agoseris	8	P
Agropyron cristatum	Crested Wheatgrass	*	P
Agrostis stolonifera	Creeping Bentgrass	*	P
Alisma gramineum	Narrowleaf Water Plantain	2	P
Alisma subcordatum	American Water Plantain	2	P
Allium stellatum	Autumn Onion	7	P
Alopecurus aequalis	Shortawn Foxtail	2	P
Alopecurus arundinaceus	Creeping Meadow Foxtail	*	P
Amaranthus albus	Prostrate Pigweed	0	A
Amaranthus retroflexus	Redroot Amaranth	0	A
Ambrosia artemisiifolia	Annual Ragweed	0	A
Ambrosia psilostachya	Cuman Ragweed	2	P
Ambrosia trifida	Great Ragweed	0	A
Amelanchier alnifolia	Saskatoon Serviceberry	6	P
Amorpha canescens	Leadplant	9	P
Amorpha fruticosa	False Indigo Bush	4	P
Andropogon gerardii	Big Bluestem	5	P
Anemone canadensis	Canadian Anemone	4	P
Anemone cylindrica	Candle Anemone	7	P
Apocynum cannabinum	Indianhemp	4	P
Arabis hirsuta	Hairy Rockcress	7	B
Arctium minus	Lesser Burdock	*	B
Artemisia absinthium	Absinthium	*	P
Artemisia biennis	Biennial Wormwood	*	B
Artemisia dracunculus	Tarragon	4	P
Artemisia frigida	Prairie Sagewort	4	P
Artemisia ludoviciana	White Sagebrush	3	P
Asclepias incarnata	Swamp Milkweed	5	P
Asclepias ovalifolia	Oval-leaf Milkweed	9	P
Asclepias speciosa	Showy Milkweed	4	P
Asclepias syriaca	Common Milkweed	0	P
Asclepias verticillata	Whorled Milkweed	3	P
Asclepias viridiflora	Green Comet Milkweed	8	P

Scientific Name[a]	Common Name[b]	C[c]	Life[d]
Asparagus officinalis	Garden Asparagus	*	P
Astragalus agrestis	Purple Milkvetch	6	P
Astragalus canadensis	Canadian Milkvetch	5	P
Astragalus flexuosus	Flexile Milkvetch	4	P
Atriplex subspicata	Saline Saltbrush	2	A
Bassia scoparia	Burningbush	*	A
Beckmannia syzigachne	American Sloughgrass	1	A
Bidens cernua	Nodding Beggartick	3	A
Bidens frondosa	Devil's Beggartick	1	A
Bidens vulgata	Big Devils Beggartick	1	A
Bolboschoenus fluviatilis	River Bulrush	2	P
Bolboschoenus maritimus	Cosmopolitan Bulrush	4	P
Boltonia asteroides	White Doll's Daisy	3	P
Bouteloua curtipendula	Sideoats Grama	5	P
Brickellia eupatorioides	False Boneset	5	P
Bromus arvensis	Field Brome	*	A
Bromus ciliatus	Fringed Brome	10	P
Bromus inermis	Smooth Brome	*	P
Calamagrostis canadensis	Bluejoint	5	P
Calamagrostis stricta	Slimstem reedgrass	5	P
Calamovilfa longifolia	Prairie Sandreed	5	P
Calylophus serrulatus	Yellow Sundrops	7	P
Calystegia sepium	Hedge False Bindweed	0	P
Campanula rotundifolia	Bluebell Bellflower	7	P
Carduus nutans	Nodding Plumeless Thistle	*	A
Carex alopecoidea	Foxtail Sedge	7	P
Carex atherodes	Wheat Sedge	4	P
Carex aurea	Golden Sedge	8	P
Carex brevior	Shortbeak Sedge	4	P
Carex cristatella	Crested Sedge	7	P
Carex granularis	Limestone Meadow Sedge	6	P
Carex gravida	Heavy Sedge	5	P
Carex lacustris	Hairy Sedge	6	P
Carex laeviconica	Smoothcone Sedge	6	P
Carex meadii	Mead's Sedge	7	P
Carex pellita	Woolly Sedge	4	P
Carex pensylvanica	Pennsylvania Sedge	8	P
Carex praegracilis	Clustered Field Sedge	5	P
Carex sartwellii	Sartwell's Sedge	5	P

Scientific Name[a]	Common Name[b]	C[c]	Life[d]
Carex stipata	Awlfruit Sedge	7	P
Carex stricta	Upright Sedge	10	P
Carex sychnocephala	Manyhead Sedge	7	P
Carex tetanica	Rigid sedge	9	P
Carex utriculata	Northwest Territory Sedge	8	P
Carex vulpinoidea	Fox Sedge	2	P
Carex xerantica	Whitescale Sedge	10	P
Cerastium arvense	Field Chickweed	2	P
Ceratophyllum demersum	Coon's Tail	4	P
Chenopodium album	Lambsquarters	*	A
Chenopodium glaucum	Oakleaf Goosefoot	*	A
Chenopodium rubrum	Red Goosefoot	2	A
Cicuta maculata	Spotted Water Hemlock	4	P
Cirsium arvense	Canada Thistle	*	P
Cirsium flodmanii	Flodman's Thistle	5	P
Cirsium vulgare	Bull Thistle	*	B
Comandra umbellata	Bastard Toadflax	8	P
Convolvulus arvensis	Field Bindweed	*	P
Conyza canadensis	Canadian Horseweed	0	A
Coreopsis tinctoria	Golden Tickseed	3	P
Cornus sericea	Redosier Dogwood	5	P
Crataegus chrysocarpa	Red Haw	6	P
Crepis runcinata	Fiddleleaf Hawksbeard	8	P
Crepis tectorum	Narrowleaf Hawksbeard	*	A
Cuscuta pentagona	Fiveangled Dodder	5	A
Cyclachaena xanthiifolia	Carelessweed	0	A
Cynoglossum officinale	Gypsyflower	*	B
Cypripedium candidum	White Lady's Slipper	10	P
Cypripedium parviflorum	Lesser Yellow Lady's Slipper	10	P
Dalea purpurea	Purple Prairie Clover	8	P
Descurainia sophia	Herb Sophia	*	A
Desmanthus illinoensis	Illinois Bundleflower	5	P
Dichanthelium leibergii	Leiberg's Panicum	8	P
Distichlis spicata	Saltgrass	2	P
Dracocephalum parviflorum	American Dragonhead	2	A
Echinacea angustifolia	Blacksamson Echinacea	7	P
Echinochloa crus-galli	Barnyardgrass	*	A
Echinochloa muricata	Rough Barnyardgrass	0	A
Elaeagnus angustifolia	Russian Olive	*	P

Scientific Name[a]	Common Name[b]	C[c]	Life[d]
Elaeagnus commutata	Silverberry	5	P
Eleocharis acicularis	Needle Spikerush	3	P
Eleocharis compressa	Flatstem Spikerush	8	P
Eleocharis palustris	Common spikerush	4	P
Elymus canadensis	Canada Wild Rye	3	P
Elymus repens	Quackgrass	*	P
Elymus trachycaulus	Slender Wheatgrass	6	P
Epilobium ciliatum	Fringed Willowherb	3	P
Epilobium leptophyllum	Bog Willowherb	6	P
Equisetum arvense	Field Horsetail	4	P
Equisetum laevigatum	Smooth Horsetail	3	P
Erechtites hieracifolia	American Burnweed	*	A
Erigeron glabellus	Streamside Fleabane	7	B
Erigeron philadelphicus	Philadelphia Fleabane	2	B
Erigeron strigosus	Prairie Fleabane	3	A
Erysimum cheiranthoides	Wormseed Wallflower	*	A
Eupatorium perfoliatum	Common Boneset	9	P
Euphorbia esula	Leafy Spurge	*	P
Euthamia graminifolia	Flat-top Goldenrod	6	P
Fragaria virginiana	Virginia Strawberry	4	P
Fraxinus pennsylvanica	Green Ash	5	P
Galium aparine	Stickywilly	0	A
Galium boreale	Northern Bedstraw	4	P
Geum aleppicum	Yellow Avens	4	P
Geum triflorum	Old Man's Whiskers	8	P
Glyceria grandis	American Mannagrass	4	P
Glyceria striata	Fowl Mannagrass	6	P
Glycyrrhiza lepidota	American Licorice	2	P
Gratiola neglecta	Clammy Hedgehyssop	0	A
Grindelia squarrosa	Curly Gumweed	1	B
Helianthus annuus	Common Sunflower	0	A
Helianthus maximiliani	Maximilian Sunflower	5	P
Helianthus nuttallii	Nuttall's Sunflower	8	P
Helianthus pauciflorus	Stiff Sunflower	8	P
Heliopsis helianthoides	Smooth Oxeye	5	P
Heliotropium curassavicum	Seaside Heliotrope	8	A
Hesperis matronalis	Dames Rocket	*	B
Hesperostipa spartea	Porcupinegrass	8	P
Heuchera richardsonii	Richardson's Alumroot	8	P

Scientific Name[a]	Common Name[b]	C[c]	Life[d]
Hierochloe odorata	Sweetgrass	10	P
Hippuris vulgaris	Common Mare's-tail	5	P
Hordeum jubatum	Foxtail Barley	0	P
Hypoxis hirsuta	Common Goldstar	8	P
Juncus arcticus	Mountain Rush	5	P
Juncus dudleyi	Dudley's Rush	4	P
Juncus interior	Inland Rush	5	P
Juncus nodosus	Knotted Rush	7	P
Juncus torreyi	Torrey's Rush	2	P
Juniperus virginiana	Eastern Redcedar	0	P
Koeleria macrantha	Prairie Junegrass	7	P
Lactuca floridana	Woodland Lettuce	4	B
Lactuca serriola	Prickly Lettuce	*	A
Lactuca tatarica	Blue Lettuce	1	P
Lathyrus palustris	Marsh Pea	9	P
Lathyrus venosus	Veiny Pea	8	P
Lemna trisulca	Star Duckweed	2	P
Lemna turionifera	Turion Duckweed	1	P
Leonurus cardiaca	Common Motherwort	*	P
Lepidium densiflorum	Common Pepperweed	0	A
Liatris ligulistylis	Rocky Mountain Blazing Star	10	P
Lilium philadelphicum	Wood Lily	8	P
Linaria vulgaris	Butter and Eggs	*	P
Linum usitatissimum	Common Flax	*	A
Lithospermum canescens	Hoary Puccoon	7	P
Lobelia spicata	Palespike Lobelia	6	P
Lonicera tatarica	Tatarian Honeysuckle	*	P
Lotus unifoliolatus	American Bird's-foot Trefoil	3	A
Lycopus americanus	American Water Horehound	4	P
Lycopus asper	Rough Bugleweed	4	P
Lysimachia ciliata	Fringed Loosestrife	6	P
Lysimachia hybrida	Lowland Yellow Loosestrife	5	P
Lysimachia thyrsiflora	Tufted Loosestrife	7	P
Maianthemum stellatum	Starry False Lily of the Valley	8	P
Medicago lupulina	Black Medick	*	P
Medicago sativa	Alfalfa	*	P
Melilotus alba	Sweetclover	*	A
Melilotus officinalis	Sweetclover	*	A
Mentha arvensis	Wild Mint	3	P

Scientific Name[a]	Common Name[b]	C[c]	Life[d]
Monarda fistulosa	Wild Bergamot	5	P
Morus alba	White Mulberry	*	P
Muhlenbergia asperifolia	Scratchgrass	2	P
Muhlenbergia racemosa	Marsh Muhly	4	P
Muhlenbergia richardsonis	Mat Muhly	10	P
Myriophyllum sibiricum	Shortspike Milfoil	3	P
Nassella Viridula	Green Needlegrass	5	P
Oenothera biennis	Common Evening Primrose	0	B
Onosmodium bejariense	Western Marbleseed	7	P
Oxalis dillenii	Slender Yellow Woodsorrel	5	P
Oxalis stricta	Common Yellow Oxalis	0	P
Oxalis violacea	Violet Woodsorrel	7	P
Packera pseudaurea	Falsegold Groundsel	5	P
Panicum capillare	Witchgrass	0	A
Panicum virgatum	Switchgrass	5	P
Parthenocissus quinquefolia	Virginia Creeper	2	P
Pascopyrum smithii	Western Wheatgrass	4	P
Pedicularis canadensis	Canadian Lousewort	10	P
Pediomelum argophyllum	Silverleaf Indian Breadroot	4	P
Pediomelum esculentum	Large Indian Breadroot	9	P
Penstemon digitalis	Foxglove Beardtongue	*	P
Phalaris arundinacea	Reed Canarygrass	0	P
Phleum pratense	Timothy	*	P
Phragmites australis	Common Reed	0	P
Physalis virginiana	Virginia Ground Cherry	4	P
Physostegia virginiana	Obedient Plant	3	P
Plantago eriopoda	Redwool Plantain	5	P
Plantago major	Common Plantain	*	P
Poa compressa	Canada Bluegrass	*	P
Poa palustris	Fowl Bluegrass	4	P
Poa pratensis	Kentucky Bluegrass	*	P
Polygonum achoreum	Leathery Knotweed	*	A
Polygonum amphibium var. emersum	Longroot Smartweed	6	P
Polygonum amphibium var. stipulaceum	Swamp Smartweed	0	P
Polygonum aviculare	Prostrate Knotweed	0	A
Polygonum convolvulus	Black Bindweed	*	A
Polygonum lapathifolia L.	Curlytop knotweed	1	A
Polygonum ramosissimum	Bushy Knotweed	3	A
Populus balsamifera	Balsam Poplar	6	P

Scientific Name[a]	Common Name[b]	C[c]	Life[d]
Populus deltoides	Eastern Cottonwood	3	P
Populus tremuloides	Quaking Aspen	4	P
Potamogeton gramineus	Variableleaf Pondweed	6	P
Potamogeton zosteriformis	Flatstem Pondweed	7	P
Potentilla anserina	Silverweed Cinquefoil	2	P
Potentilla arguta	Tall Cinquefoil	8	P
Potentilla gracilis	Slender Cinquefoil	5	P
Potentilla norvegica	Norwegian Cinquefoil	0	A
Potentilla pensylvanica	Pennsylvania Cinquefoil	9	P
Potentilla rivalis	Brook Conquefoil	3	A
Prenanthes racemosa	Purple Rattlesnakeroot	10	P
Prunus americana	American Plum	4	P
Prunus virginiana	Chokecherry	4	P
Puccinellia nuttalliana	Nuttall's Alkaligrass	4	P
Ranunculus cymbalaria	Alkali Buttercup	3	P
Ranunculus flabellaris	Yellow Water Buttercup	7	P
Ranunculus gmelinii	Gmelin's Buttercup	8	P
Ranunculus longirostris	Longbeak Buttercup	7	P
Ranunculus macounii	Macoun's Buttercup	4	A
Ranunculus pensylvanicus	Pennsylvania Buttercup	4	A
Ranunculus sceleratus	Cursed Buttercup	3	A
Ratibida columnifera	Upright Prairie Coneflower	3	P
Ratibida pinnata	Pinnate Prairie Coneflower	6	P
Rhamnus cathartica	Common Buckthorn	*	P
Ribes americanum	American Black Currant	7	P
Rorippa palustris	Bog Yellowcress	2	A
Rorippa sinuata	Spreading Yellowcress	4	P
Rosa arkansana	Prairie Rose	3	P
Rosa woodsii	Wood's Rose	5	P
Rubus idaeus	American Red Raspberry	5	P
Rudbeckia hirta	Blackeyed Susan	5	B
Rumex aquaticus	Western Dock	7	P
Rumex crispus	Curly Dock	*	P
Rumex maritimus	Golden Dock	1	A
Rumex salicifolius	Mexican Dock	1	P
Salicornia rubra	Red Swampfire	0	A
Salix amygdaloides	Peachleaf Willow	3	P
Salix bebbiana	Bebb Willow	8	P
Salix eriocephala	Missouri River Willow	5	P

Scientific Name[a]	Common Name[b]	C[c]	Life[d]
Salix Interior	Sandbar Willow	3	P
Salix petiolaris	Meadow Willow	8	P
Salsola tragus	Prickly Russian Thistle	*	A
Schizachyrium scoparium	Little Bluestem	6	P
Schoenoplectus acutus	Hardstem Bulrush	5	P
Schoenoplectus pungens	Common Threesquare	4	P
Schoenoplectus tabernaemontani	Softstem Bulrush	3	P
Scirpus pallidus	Cloaked Bulrush	5	P
Scolochloa festucacea	Common Rivergrass	6	P
Scutellaria galericulata	Marsh Skullcap	7	P
Scutellaria lateriflora	Blue Skullcap	6	P
Securigera varia	Crownvetch	*	P
Senecio congestus	Marsh Fleabane	2	A
Shepherdia argentea	Silver Buffaloberry	5	P
Silene noctiflora	Nightflowering Silene	*	A
Silphium laciniatum	Compass Plant	8	P
Sinapis arvensis	Wild Mustard	*	A
Sisyrinchium campestre	Prairie Blue-eyed Grass	10	P
Sium suave	Hemlock Waterparsnip	3	P
Solanum dulcamara	Climbing Nightshade	*	P
Solidago canadensis	Canada Goldenrod	1	P
Solidago gigantea	Giant Goldenrod	4	P
Solidago missouriensis	Missouri Goldenrod	5	P
Solidago mollis	Velvety Goldenrod	6	P
Solidago rigida	Stiff Goldenrod	4	P
Sonchus arvensis	Field Sowthistle	*	P
Sorghastrum nutans	Indiangrass	6	P
Sparganium eurycarpum	Broadfruit Bur-reed	4	P
Spartina gracilis	Alkali Cordgrass	6	P
Spartina pectinata	Prairie Cordgrass	5	P
Spiraea alba	White Meadowsweet	7	P
Sporobolus heterolepis	Prairie Dropseed	10	P
Stachys pilosa	Hairy Hedgenettle	3	P
Stellaria longifolia	Longleaf Starwort	8	P
Stuckenia pectinata	Sago Pondweed	0	P
Suaeda calceoliformis	Pursh Seepweed	2	A
Symphoricarpos occidentalis	Western Snowberry	3	P
Symphyotrichum ciliatum	Rayless Alkali Aster	0	A
Symphyotrichum ericoides	White Heath Aster	2	P

Scientific Name[a]	Common Name[b]	C[c]	Life[d]
Symphyotrichum falcatum	White Prairie Aster	4	P
Symphyotrichum laeve	Smooth Blue Aster	5	P
Symphyotrichum lanceolatum	White Panicle Aster	3	P
Tanacetum vulgare	Common Tansy	*	P
Taraxacum officinale	Common Dandelion	*	P
Teucrium canadense	Canada Germander	3	P
Thalictrum dasycarpum	Purple Meadow-rue	7	P
Thalictrum venulosum	Veiny Meadow-rue	6	P
Thermopsis rhombifolia	Prairie Thermopsis	6	P
Thinopyrum intermedium	Intermediate Wheatgrass	*	P
Thlaspi arvense	Field Pennycress	*	A
Toxicodendron rydbergii	Western Poison Ivy	3	P
Tradescantia bracteata	Longbract Spiderwort	7	P
Tragopogon dubius	Yellow Salsify	*	B
Trifolium hybridum	Alsike Clover	*	P
Trifolium pratense	Red Clover	*	P
Trifolium repens	White Clover	*	P
Triglochin maritima	Seaside Arrowgrass	5	P
Tripleurospermum perforatum	Scentless False Mayweed	*	A
Typha angustifolia	Narrowleaf Cattail	*	P
Typha latifolia	Broadleaf Cattail	2	P
Typha x glauca	Hybrid Cattail	*	P
Ulmus americana	American Elm	3	P
Ulmus pumila	Siberian Elm	*	P
Urtica dioica	Stinging Nettle	0	P
Utricularia macrorhiza	Common Bladderwort	2	P
Verbena hastata	Swamp Verbena	5	P
Verbena stricta	Hoary Verbena	2	P
Vernonia fasciculata	Prairie Ironweed	3	P
Veronica peregrina	Neckweed	0	A
Vicia americana	American Vetch	6	P
Viola nephrophylla	Northern Bog Violet	8	P
Viola nuttallii	Nutall's Violet	8	P
Viola pedatifida	Prairie Violet	8	P
Viola pubescens	Downy Yellow Violet	9	P
Xanthium strumarium	Rough Cocklebur	0	A
Zannichellia palustris	Horned Pondweed	2	P
Zigadenus elegans	Mountain Deathcamas	8	P
Zizia aptera	Meadow Zizia	8	P

Scientific Name[a]	Common Name[b]	C[c]	Life[d]
Zizia aurea	Golden Zizia	8	P

[a]Species scientific names follow the nomenclature of the USDA Plants Database (USDA. NRCS 2011).
[b]Common names follow the nomenclature of the USDA Plants Database (USDA. NRCS 2011).
[c]Coefficient of Conservatism (NGPFQAP 2001).
[d]Life history guild – P = Perennial. A = Annual. B = Biennial.
*Introduced species are not assigned a coefficient of conservatism.

APPENDIX B. WETLAND SITES AND SITE CHARACTERISTICS

Wetland ID[a]	Year[b]	Grassland[c]	Class[d]	IPCI condition[e]	IPCI Score[f]	NDRAM Condition[g]	NDRAM Score[h]
92	Year1	Reseeded	Seasonal	Fair	57	Fair High	60
140	Year2	Native	Temporary	Fair	34	Fair high	63
352	Year2	Reseeded	Seasonal	Poor	23	Fair low	41
490	Year2	Reseeded	Seasonal	Fair	52	Fair low	48
517	Year2	Reseeded	Seasonal	Poor	20	Fair low	41
584	Year1	Native	Seasonal	Very Good	87	Fair High	60
659	Year2	Native	Seasonal	Poor	37	Fair high	53
802	Year1	Reseeded	Seasonal	Poor	31	Fair High	58
842	Year2	Reseeded	Seasonal	Poor	38	Fair high	56
844	Year1	Reseeded	Seasonal	Fair	53	Fair low	51
1019	Year2	Reseeded	Seasonal	Poor	24	Fair low	39
1056	Year2	Reseeded	Seasonal	Poor	31	Fair low	39
1150	OverSamp	Reseeded	Seasonal	Poor	24	Fair low	51
1204	OverSamp	Reseeded	Temporary	Poor	0	Fair High	53
1265	Year2	Reseeded	Seasonal	Poor	27	Fair low	41
1273	Year1	Reseeded	Seasonal	Fair	50	Fair High	53
1432	Year2	Reseeded	Seasonal	Fair	55	Fair high	53
1629	Year2	Reseeded	Seasonal	Poor	34	Fair low	41
1689	Year1	Reseeded	Temporary	Poor	16	Fair High	60
1755	Year2	Reseeded	Seasonal	Good	68	Fair high	53
1772	Year2	Reseeded	Seasonal	Very poor	12	Fair low	48
1969	Year1	Reseeded	Seasonal	Fair	44	Fair High	55
2390	OverSamp	Reseeded	Temporary	Poor	0	Fair High	55
2473	Year2	Reseeded	Seasonal	Fair	57	Fair high	53
2576	OverSamp	Reseeded	Seasonal	Fair	50	Fair High	60
2629	Year2	Reseeded	Seasonal	Good	65	Fair high	53
2729	Year2	Reseeded	Seasonal	Very poor	4	Fair low	41
2818	Year2	Reseeded	Seasonal	Poor	36	Fair high	60
3064	Year1	Native	Seasonal	Poor	35	Fair High	56
3144	Year2	Reseeded	Seasonal	Poor	37	Fair high	56
3275	Year1	Reseeded	Seasonal	Fair	44	Fair High	56
3476	Year2	Reseeded	Seasonal	Good	69	Fair high	65
3500	OverSamp	Reseeded	Seasonal	Fair	41	Fair High	56
3572	Year1	Reseeded	Seasonal	Very Poor	4	Fair Low	34
3762	OverSamp	Reseeded	Seasonal	Poor	35	Fair High	62
3798	OverSamp	Reseeded	Temporary	Poor	15	Fair Low	46

Wetland ID[a]	Year[b]	Grassland[c]	Class[d]	IPCI condition[e]	IPCI Score[f]	NDRAM Condition[g]	NDRAM Score[h]
3890	Year1	Reseeded	Seasonal	Poor	30	Fair Low	51
4096	Year1	Reseeded	Seasonal	Poor	20	Fair High	53
4130	Year1	Reseeded	Temporary	Poor	16	Fair High	54
4189	Year2	Reseeded	Seasonal	Fair	41	Fair high	65
4226	Year2	Reseeded	Seasonal	Poor	28	Fair low	51
4307	Year2	Reseeded	Seasonal	Poor	32	Fair high	54
4488	Year1	Reseeded	Temporary	Fair	52	Fair High	64
4490	Year1	Reseeded	Seasonal	Good	65	Fair High	58
4548	Year2	Reseeded	Temporary	Fair	64	Fair high	65
4587	OverSamp	Native	Seasonal	Very good	91	Good	72
4698	Year1	Reseeded	Temporary	Poor	4	Fair Low	50
4802	Year1	Reseeded	Seasonal	Poor	0	Fair Low	51
4873	OverSamp	Reseeded	Temporary	Poor	27	Fair High	58
4976	Year2	Reseeded	Seasonal	Poor	24	Fair low	41
5130	Year1	Reseeded	Seasonal	Poor	35	Fair High	53
5133	Year2	Reseeded	Seasonal	Good	69	Fair high	65
5177	Year2	Reseeded	Seasonal	Poor	24	Fair low	44
5298	Year1	Reseeded	Seasonal	Poor	32	Fair High	53
5327	Year1	Reseeded	Seasonal	Good	61	Fair High	57
5791	Year2	Reseeded	Seasonal	Poor	32	Fair low	41
5813	OverSamp	Reseeded	Seasonal	Poor	24	Fair low	41
5865	Year1	Reseeded	Seasonal	Fair	55	Fair Low	45
6019	Year2	Reseeded	Seasonal	Very poor	0	Fair low	41
6230	Year2	Reseeded	Temporary	Poor	15	Fair high	57
6243	Year1	Native	Seasonal	Very good	80	Good	82
6261	OverSamp	Reseeded	Seasonal	Poor	32	Fair low	51
6300	Year2	Reseeded	Seasonal	Poor	24	Fair low	41
6316	Year1	Reseeded	Seasonal	Poor	20	Fair High	55
6433	Year1	Reseeded	Seasonal	Good	68	Fair High	60
6547	OverSamp	Reseeded	Temporary	Poor	0	Fair high	54
6576	Year2	Reseeded	Seasonal	Poor	32	Fair low	48
6578	Year1	Reseeded	Seasonal	Good	69	Fair High	62
6666	Year1	Reseeded	Temporary	Fair	55	Good	71
7273	OverSamp	Native	Seasonal	Good	64	Fair High	59
7312	Year1	Reseeded	Temporary	Fair	55	Fair High	53
7481	Year2	Reseeded	Temporary	Fair	38	Fair high	59
7498	Year1	Reseeded	Seasonal	Poor	34	Fair High	53

Wetland ID[a]	Year[b]	Grassland[c]	Class[d]	IPCI condition[e]	IPCI Score[f]	NDRAM Condition[g]	NDRAM Score[h]
7509	Year2	Reseeded	Seasonal	Fair	42	Fair low	51
7583	OverSamp	Native	Seasonal	Good	79	Fair High	65
7604	Year1	Native	Seasonal	Good	75	Good	86
7683	Year1	Reseeded	Seasonal	Good	73	Fair High	60
8217	Year1	Reseeded	Seasonal	Fair	49	Fair High	58
8307	OverSamp	Native	Seasonal	Good	62	Good	70
8371	Year2	Native	Seasonal	Good	71	Good	86
8388	Year1	Native	Seasonal	Good	65	Good	83
8414	Year1	Native	Seasonal	Good	76	Good	84
8433	Year2	Native	Seasonal	Good	72	Good	86
8434	Year2	Native	Temporary	Good	76	Good	86
8441	Year2	Native	Seasonal	Good	69	Good	81
8520	OverSamp	Native	Seasonal	Good	69	Good	83
8681	Year2	Reseeded	Seasonal	Very poor	7	Fair low	44
8737	Year2	Reseeded	Seasonal	Fair	49	Fair high	54
8758	Year2	Reseeded	Seasonal	Poor	24	Fair low	41
8871	Year2	Reseeded	Seasonal	Very poor	0	Fair low	41
8918	Year2	Reseeded	Temporary	Fair	35	Fair high	53
8958	Year1	Native	Seasonal	Good	73	Good	86
9121	Year1	Native	Seasonal	Good	61	Good	81
9378	Year2	Native	Temporary	Fair	50	Good	74
9522	Year1	Reseeded	Seasonal	Very Poor	4	Fair Low	40
9615	Year2	Reseeded	Seasonal	Poor	23	Fair high	56
9653	OverSamp	Reseeded	Seasonal	Fair	41	Fair low	46
9826	Year2	Reseeded	Seasonal	Poor	31	Fair high	53
10453	Year2	Reseeded	Seasonal	Fair	53	Fair high	65
10513	OverSamp	Reseeded	Temporary	Fair	45	Fair high	56
10674	Year1	Native	Temporary	Poor	26	Fair Low	44
10941	OverSamp	Reseeded	Seasonal	Fair	57	Fair high	56
11087	Year1	Reseeded	Seasonal	Poor	28	Fair High	54
11090	Year2	Reseeded	Seasonal	Very poor	16	Fair low	49
11091	Year2	Reseeded	Seasonal	Poor	31	Fair high	54
11121	Year2	Reseeded	Seasonal	Poor	28	Fair high	56
11181	Year2	Reseeded	Seasonal	Fair	50	Fair high	56
11435	Year2	Reseeded	Temporary	Poor	12	Fair low	41
11470	Year2	Native	Seasonal	Very Good	80	Good	78
11471	Year2	Reseeded	Temporary	Poor	8	Fair low	37

Wetland ID[a]	Year[b]	Grassland[c]	Class[d]	IPCI condition[e]	IPCI Score[f]	NDRAM Condition[g]	NDRAM Score[h]
11542	OverSamp	Native	Temporary	Fair	60	Fair high	65
11605	Year1	Reseeded	Temporary	Poor	4	Fair low	51
12091	Year2	Reseeded	Seasonal	Poor	20	Fair low	49
12101	Year2	Reseeded	Temporary	Poor	31	Fair low	49
12199	Year2	Reseeded	Seasonal	Very poor	0	Fair low	41
12445	OverSamp	Reseeded	Seasonal	Good	65	Fair high	56
12714	Year1	Reseeded	Seasonal	Good	65	Fair High	53
12788	Year1	Reseeded	Seasonal	Good	65	Fair High	60
12866	Year1	Native	Seasonal	Good	62	Fair High	66
12933	Year1	Native	Temporary	Good	83	Good	75
13004	Year1	Reseeded	Seasonal	Good	61	Fair High	60
13221	Year1	Reseeded	Temporary	Very Poor	8	Fair Low	49
13270	Year1	Reseeded	Seasonal	Very good	80	Fair High	60
13285	Year2	Native	Temporary	Fair	51	Fair high	59
13422	Year2	Reseeded	Seasonal	Fair	57	Fair high	60
13468	Year2	Reseeded	Seasonal	Poor	28	Fair low	51
13612	Year1	Reseeded	Temporary	Fair	45	Fair High	60
13616	OverSamp	Reseeded	Temporary	Fair	44	Fair high	60
13621	Year2	Reseeded	Temporary	Fair	35	Fair high	60
13634	Year1	Reseeded	Seasonal	Good	62	Fair High	60
13798	Year1	Reseeded	Seasonal	Good	61	Fair High	60
13873	Year1	Reseeded	Temporary	Poor	26	Fair High	58
14018	Year1	Reseeded	Temporary	Fair	45	Fair High	60
14026	Year2	Reseeded	Seasonal	Poor	35	Fair low	51
14050	Year2	Native	Seasonal	Very Good	87	Good	80
14088	Year1	Reseeded	Seasonal	Poor	38	Fair low	54
14590	Year2	Reseeded	Temporary	Poor	8	Fair low	41
14749	Year1	Native	Seasonal	Poor	28	Fair High	65
14798	Year2	Reseeded	Temporary	Poor	20	Fair high	53
14988	Year2	Native	Seasonal	Good	60	Good	70
15026	Year2	Reseeded	Seasonal	Fair	47	Fair high	62
15156	Year1	Reseeded	Temporary	Poor	19	Fair low	51
15324	Year2	Reseeded	Seasonal	Good	60	Fair high	60
15403	OverSamp	Reseeded	Seasonal	Fair	54	Fair high	54
15626	OverSamp	Native	Temporary	Fair	59	Good	82
15629	Year1	Reseeded	Seasonal	Very Poor	8	Fair Low	51
15675	Year1	Reseeded	Seasonal	Fair	44	Fair low	45

Wetland ID[a]	Year[b]	Grassland[c]	Class[d]	IPCI condition[e]	IPCI Score[f]	NDRAM Condition[g]	NDRAM Score[h]
15823	Year2	Native	Seasonal	Very Good	81	Good	85
15880	Year2	Reseeded	Seasonal	Fair	44	Fair low	51
16023	Year2	Reseeded	Seasonal	Very poor	15	Fair low	49
16143	Year1	Native	Seasonal	Very good	99	Good	91
16188	Year1	Reseeded	Seasonal	Fair	44	Fair High	58
16198	Year1	Reseeded	Seasonal	Poor	38	Fair low	49
16389	Year2	Reseeded	Seasonal	Poor	38	Fair high	65
16607	Year1	Native	Seasonal	Good	73	Fair High	67
16776	Year2	Reseeded	Temporary	Poor	0	Fair low	41
16871	Year1	Reseeded	Temporary	Poor	19	Fair High	53
16888	Year1	Reseeded	Seasonal	Poor	34	Fair High	58
17193	Year1	Native	Seasonal	Very good	83	Good	82
17463	Year1	Native	Seasonal	Very good	91	Good	79
17695	Year1	Reseeded	Seasonal	Poor	27	Fair High	53
17844	Year2	Reseeded	Temporary	Poor	8	Fair high	56
17858	Year1	Reseeded	Temporary	Poor	0	Fair low	51
17917	Year1	Native	Temporary	Fair	35	Good	70
17966	Year1	Native	Seasonal	Very good	99	Good	98
18110	Year2	Reseeded	Temporary	Poor	0	Fair low	38
18116	Year1	Reseeded	Temporary	Poor	16	Fair High	58
18331	Year1	Native	Seasonal	Good	79	Good	80
18470	Year1	Native	Seasonal	Very good	26	Good	58
18499	Year1	Reseeded	Temporary	Poor	99	Fair High	72
18527	Year1	Native	Seasonal	Very good	91	Good	58
18682	Year1	Native	Seasonal	Very good	95	Fair High	78
18843	Year2	Reseeded	Seasonal	Poor	27	Fair low	47
19162	Year1	Reseeded	Temporary	Poor	4	Fair High	53
19180	Year2	Reseeded	Temporary	Poor	4	Fair low	44
19263	Year1	Native	Seasonal	Very good	91	Good	70
19410	Year1	Reseeded	Seasonal	Poor	23	Fair Low	51
19660	Year1	Reseeded	Seasonal	Poor	35	Fair High	54
19733	Year2	Reseeded	Seasonal	Good	72	Fair high	63
19839	Year1	Reseeded	Temporary	Fair	65	Fair High	60
20166	Year2	Reseeded	Temporary	Poor	31	Fair high	60
20315	Year1	Reseeded	Seasonal	Fair	58	Fair High	60
20436	Year1	Reseeded	Seasonal	Poor	35	Fair High	60
20870	Year1	Reseeded	Temporary	Poor	0	Fair Low	48

Wetland ID[a]	Year[b]	Grassland[c]	Class[d]	IPCI condition[e]	IPCI Score[f]	NDRAM Condition[g]	NDRAM Score[h]
20945	Year2	Reseeded	Seasonal	Fair	54	Fair low	49
20984	Year2	Native	Seasonal	Good	60	Good	72
21153	Year2	Reseeded	Seasonal	Very poor	4	Fair low	41
21222	Year1	Reseeded	Seasonal	Poor	24	Fair High	62
21290	Year1	Native	Temporary	Fair	51	Fair High	68
21307	Year2	Native	Temporary	Poor	8	Good	70
21386	Year2	Reseeded	Temporary	Fair	35	Fair low	49
21509	Year2	Native	Temporary	Fair	34	Fair high	61
21542	Year1	Reseeded	Temporary	Poor	7	Fair Low	42
21720	Year1	Native	Temporary	Poor	12	Fair High	58
21779	Year1	Reseeded	Temporary	Fair	39	Fair High	58
22040	Year2	Reseeded	Temporary	Fair	39	Fair low	46
22252	Year1	Native	Seasonal	Poor	23	Fair High	53
22415	Year2	Reseeded	Temporary	Poor	16	Fair low	44
22427	Year2	Reseeded	Seasonal	Poor	28	Fair low	44
22737	Year2	Reseeded	Seasonal	Very poor	12	Fair low	44

[a] Wetland identification number assigned by the FWS
[b] Designated year the wetland was supposed to be assessed (Oversample = taken from the oversample population)
[c] Type of grassland the wetland was located in
[d] Wetland Classification (Steward and Kantrud 1971)
[e] Index of Plant Community Integrity condition rating
[f] Index of Plant Community Integrity condition score
[g] North Dakota Rapid Assessment Method condition rating
[h] North Dakota Rapid Assessment Method condition score

APPENDIX C. PRIMARY SPECIES TRIMMED AND RETAINED FOR NMS ORDINATION

Temporary Wetlands Species Trimmed	
Low Prairie	**Wet Meadow**
Agoseris glauca	*Achillea millefolium*
Agrostis stolonifera	*Agrostis stolonifera*
Alopecurus arundinaceus	*Alopecurus aequalis*
Asclepias ovalifolia	*Ambrosia artemisiifolia*
Astragalus agrestis	*Ambrosia trifida*
Astragalus canadensis	*Artemisia ludoviciana*
Atriplex subspicata	*Beckmannia syzigachne*
Calamagrostis stricta	*Bolboschoenus fluviatilis*
Cirsium vulgare	*Bromus ciliatus*
Cornus sericea	*Chenopodium glaucum*
Carex aurea	*Chenopodium rubrum*
Carex pensylvanica	*Cirsium vulgare*
Carex tetanica	*Convolvulus arvensis*
Dalea purpurea	*Carex praegracilis*
Distichlis spicata	*Carex sartwellii*
Eleocharis palustris	*Carex stricta*
Equisetum arvense	*Carex sychnocephala*
Equisetum laevigatum	*Carex vulpinoidea*
Erysimum cheiranthoides	*Descurainia sophia*
Fraxinus pennsylvanica	*Distichlis spicata*
Galium aparine	*Eleocharis acicularis*
Helianthus annuus	*Equisetum arvense*
Heliopsis helianthoides	*Equisetum laevigatum*
Helianthus nuttallii	*Erysimum cheiranthoides*
Heuchera richardsonii	*Euphorbia esula*
Juncus arcticus	*Fraxinus pennsylvanica*
Lactuca floridana	*Galium aparine*
Lactuca serriola	*Galium boreale*
Liatris ligulistylis	*Glyceria striata*
Lonicera tatarica	*Helianthus annuus*
Lycopus americanus	*Helianthus maximiliani*
Lycopus asper	*Helianthus nuttallii*
Medicago sativa	*Helianthus pauciflorus*
Oenothera biennis	*Hierochloe odorata*
Oxalis dillenii	*Lactuca floridana*
Oxalis stricta	*Lactuca serriola*
Oxalis violacea	*Lactuca tatarica*
Panicum virgatum	*Lotus unifoliolatus*
Polygonum lapathifolia L.	*Lycopus asper*
Plantago major	*Mentha arvensis*

Temporary Wetlands Species Trimmed	
Low Prairie	**Wet Meadow**
Poa palustris	*Muhlenbergia asperifolia*
Populus balsamifera	*Plantago major*
Populus tremuloides	*Polygonum convolvulus*
Prunus americana	*Polygonum ramosissimum*
Prunus virginiana	*Populus tremuloides*
Rubus idaeus	*Prunus virginiana*
Rudbeckia hirta	*Ranunculus macounii*
Salix petiolaris	*Rorippa palustris*
Schizachyrium scoparium	*Rumex salicifolius*
Solidago missouriensis	*Salix amygdaloides*
Spiraea alba	*Salix Interior*
Stellaria longifolia	*Salicornia rubra*
Symphyotrichum ciliatum	*Salsola tragus*
Symphyotrichum falcatum	*Schoenoplectus acutus*
Thalictrum venulosum	*Schoenoplectus pungens*
Thinopyrum intermedium	*Solidago gigantea*
Tradescantia bracteata	*Stellaria longifolia*
Trifolium pratense	*Symphyotrichum ciliatum*
Trifolium repens	*Symphyotrichum ericoides*
Urtica dioica	*Thlaspi arvense*
Vernonia fasciculata	*Trifolium pratense*
Viola nephrophylla	*Typha angustifolia*
Viola pedatifida	*Typha x glauca*
	Urtica dioica
	Utricularia macrorhiza
	Veronica peregrina
	Vicia americana
	Xanthium strumarium
	Zizia aurea

Temporary Wetlands Species Retained	
Low Prairie	**Wet Meadow**
Achillea millefolium	*Alopecurus arundinaceus*
Agropyron cristatum	*Ambrosia psilostachya*
Ambrosia artemisiifolia	*Ambrosia trifida*
Ambrosia psilostachya	*Apocynum cannabinum*
Amorpha canescens	*Artemisia absinthium*
Andropogon gerardii	*Asclepias speciosa*
Anemone canadensis	*Atriplex subspicata*
Apocynum cannabinum	*Bromus inermis*
Artemisia absinthium	*Calamagrostis canadensis*
Artemisia ludoviciana	*Calamagrostis stricta*
Asclepias speciosa	*Chenopodium album*
Bromus inermis	*Cirsium arvense*
Cirsium arvense	*Carex atherodes*
Cirsium flodmanii	*Carex brevior*
Comandra umbellata	*Carex laeviconica*
Convolvulus arvensis	*Carex pellita*
Carex brevior	*Eleocharis palustris*
Carex laeviconica	*Elymus repens*
Carex pellita	*Glycyrrhiza lepidota*
Carex praegracilis	*Hordeum jubatum*
Elaeagnus commutata	*Juncus arcticus*
Elymus repens	*Lycopus americanus*
Elymus trachycaulus	*Medicago lupulina*
Euphorbia esula	*Melilotus officinalis*
Galium boreale	*Pascopyrum smithii*
Glycyrrhiza lepidota	*Polygonum amphibium var. stipulaceum*
Grindelia squarrosa	*Polygonum lapathifolia L.*
Helianthus maximiliani	*Phalaris arundinacea*
Helianthus pauciflorus	*Phleum pratense*
Hordeum jubatum	*Poa palustris*
Lactuca tatarica	*Poa pratensis*
Medicago lupulina	*Potentilla anserina*
Melilotus officinalis	*Potentilla norvegica*
Monarda fistulosa	*Rosa woodsii*
Nassella Viridula	*Rumex crispus*
Pascopyrum smithii	*Solidago canadensis*
Polygonum amphibium var. stipulaceum	*Sonchus arvensis*
Phalaris arundinacea	*Spartina pectinata*
Phleum pratense	*Stachys pilosa*
Poa pratensis	*Symphyotrichum lanceolatum*
Potentilla anserina	*Symphoricarpos occidentalis*
Potentilla norvegica	*Taraxacum officinale*
Pediomelum argophyllum	*Teucrium canadense*
Ratibida columnifera	

Temporary Wetlands Species Retained	
Low Prairie	**Wet Meadow**
Rosa arkansana	
Rosa woodsii	
Rumex crispus	
Solidago canadensis	
Solidago gigantea	
Solidago rigida	
Sonchus arvensis	
Sorghastrum nutans	
Spartina pectinata	
Stachys pilosa	
Symphyotrichum ericoides	
Symphyotrichum lanceolatum	
Symphoricarpos occidentalis	
Taraxacum officinale	
Teucrium canadense	
Tragopogon dubius	
Vicia americana	
Zizia aptera	
Zizia aurea	

Seasonal Wetlands Species Trimmed		
Low Prairie	**Wet Meadow**	**Shallow Marsh**
Acer negundo	*Acer negundo*	*Alisma gramineum*
Agoseris glauca	*Achillea millefolium*	*Alisma subcordatum*
Agrostis stolonifera	*Agrostis stolonifera*	*Alopecurus arundinaceus*
Allium stellatum	*Alisma subcordatum*	*Ambrosia artemisiifolia*
Alopecurus arundinaceus	*Ambrosia artemisiifolia*	*Apocynum cannabinum*
Ambrosia artemisiifolia	*Andropogon gerardii*	*Artemisia absinthium*
Anemone cylindrica	*Artemisia biennis*	*Artemisia biennis*
Arctium minus	*Artemisia ludoviciana*	*Atriplex subspicata*
Asclepias syriaca	*Asclepias incarnata*	*Beckmannia syzigachne*
Asclepias verticillata	*Astragalus canadensis*	*Bidens cernua*
Asclepias viridiflora	*Atriplex subspicata*	*Bidens frondosa*
Astragalus agrestis	*Beckmannia syzigachne*	*Bidens vulgata*
Atriplex subspicata	*Bidens cernua*	*Bolboschoenus maritimus*
Bassia scoparia	*Bidens frondosa*	*Boltonia asteroides*
Boltonia asteroides	*Bidens vulgata*	*Calamagrostis canadensis*
Bouteloua curtipendula	*Bolboschoenus fluviatilis*	*Calamagrostis stricta*
Brickellia eupatorioides	*Calystegia sepium*	*Calystegia sepium*
Calamagrostis canadensis	*Chenopodium rubrum*	*Ceratophyllum demersum*
Calystegia sepium	*Cicuta maculata*	*Chenopodium glaucum*
Cerastium arvense	*Cirsium flodmanii*	*Cicuta maculata*
Chenopodium album	*Cirsium vulgare*	*Cirsium vulgare*
Chenopodium glaucum	*Conyza canadensis*	*Conyza canadensis*
Chenopodium rubrum	*Cornus sericea*	*Carex lacustris*
Cicuta maculata	*Carex cristatella*	*Carex sartwellii*
Cirsium vulgare	*Carex granularis*	*Carex stricta*
Comandra umbellata	*Carex lacustris*	*Carex utriculata*
Conyza canadensis	*Carex praegracilis*	*Distichlis spicata*
Crataegus chrysocarpa	*Carex stricta*	*Echinochloa crus-galli*
Carex atherodes	*Carex sychnocephala*	*Eleocharis acicularis*
Carex cristatella	*Carex tetanica*	*Elymus repens*
Carex granularis	*Carex utriculata*	*Equisetum arvense*
Carex meadii	*Cynoglossum officinale*	*Euphorbia esula*
Carex praegracilis	*Distichlis spicata*	*Fraxinus pennsylvanica*
Carex sartwellii	*Echinochloa crus-galli*	*Glyceria grandis*
Carex tetanica	*Elaeagnus angustifolia*	*Glycyrrhiza lepidota*
Carex vulpinoidea	*Eleocharis compressa*	*Helianthus annuus*
Carex xerantica	*Erigeron philadelphicus*	*Helianthus nuttallii*
Cynoglossum officinale	*Erysimum cheiranthoides*	*Hippuris vulgaris*
Cypripedium candidum	*Eupatorium perfoliatum*	*Juncus arcticus*
Dalea purpurea	*Euthamia graminifolia*	*Lysimachia hybrida*
Desmanthus illinoensis	*Fraxinus pennsylvanica*	*Melilotus officinalis*
Dichanthelium leibergii	*Fragaria virginiana*	*Mentha arvensis*
Distichlis spicata	*Galium boreale*	*Muhlenbergia asperifolia*
Dracocephalum parviflorum	*Glyceria striata*	*Myriophyllum sibiricum*

Seasonal Wetlands Species Trimmed		
Low Prairie	**Wet Meadow**	**Shallow Marsh**
Eleocharis compressa	*Helianthus annuus*	*Pascopyrum smithii*
Eleocharis palustris	*Heliopsis helianthoides*	*Polygonum amphibium var. emersum*
Elymus canadensis	*Helianthus pauciflorus*	*Phleum pratense*
Elymus trachycaulus	*Hierochloe odorata*	*Polygonum convolvulus*
Equisetum arvense	*Hypoxis hirsuta*	*Polygonum ramosissimum*
Erigeron glabellus	*Juncus dudleyi*	*Potentilla anserina*
Erigeron philadelphicus	*Juncus interior*	*Potamogeton gramineus*
Eupatorium perfoliatum	*Juncus torreyi*	*Puccinellia nuttalliana*
Fraxinus pennsylvanica	*Lactuca tatarica*	*Ranunculus cymbalaria*
Fragaria virginiana	*Lepidium densiflorum*	*Ranunculus sceleratus*
Geum aleppicum	*Linaria vulgaris*	*Rorippa palustris*
Glyceria striata	*Lysimachia ciliata*	*Rumex maritimus*
Grindelia squarrosa	*Lysimachia hybrida*	*Rumex salicifolius*
Helianthus annuus	*Medicago lupulina*	*Salix Interior*
Hesperostipa spartea	*Medicago sativa*	*Schoenoplectus pungens*
Heuchera richardsonii	*Muhlenbergia asperifolia*	*Schoenoplectus tabernaemontani*
Juncus dudleyi	*Muhlenbergia richardsonis*	*Scirpus pallidus*
Juniperus virginiana	*Oenothera biennis*	*Scolochloa festucacea*
Lactuca serriola	*Oxalis stricta*	*Sium suave*
Lathyrus palustris	*Oxalis violacea*	*Solidago canadensis*
Liatris ligulistylis	*Packera pseudaurea*	*Spartina gracilis*
Lilium philadelphicum	*Panicum virgatum*	*Stachys pilosa*
Linaria vulgaris	*Pascopyrum smithii*	*Stuckenia pectinata*
Lithospermum canescens	*Polygonum amphibium var. emersum*	*Symphyotrichum lanceolatum*
Lotus unifoliolatus	*Phleum pratense*	*Tanacetum vulgare*
Lycopus americanus	*Phragmites australis*	*Thlaspi arvense*
Lycopus asper	*Plantago major*	*Toxicodendron rydbergii*
Maianthemum stellatum	*Polygonum convolvulus*	*Tragopogon dubius*
Mentha arvensis	*Polygonum ramosissimum*	*Typha latifolia*
Muhlenbergia richardsonis	*Populus deltoides*	*Urtica dioica*
Nassella Viridula	*Potentilla rivalis*	*Vicia americana*
Oxalis stricta	*Puccinellia nuttalliana*	*Xanthium strumarium*
Oxalis violacea	*Ranunculus macounii*	*Zannichellia palustris*
Packera pseudaurea	*Ranunculus pensylvanicus*	
Panicum virgatum	*Rorippa palustris*	
Pascopyrum smithii	*Rorippa sinuata*	
Physalis virginiana	*Rosa arkansana*	
Plantago major	*Rumex salicifolius*	
Poa palustris	*Salix amygdaloides*	
Polygonum convolvulus	*Salix eriocephala*	

Seasonal Wetlands Species Trimmed		
Low Prairie	**Wet Meadow**	**Shallow Marsh**
Populus tremuloides	*Salix petiolaris*	
Potentilla arguta	*Salsola tragus*	
Potentilla norvegica	*Scirpus pallidus*	
Prenanthes racemosa	*Scolochloa festucacea*	
Prunus virginiana	*Sium suave*	
Ratibida columnifera	*Sparganium eurycarpum*	
Ratibida pinnata	*Spartina gracilis*	
Rhamnus cathartica	*Spiraea alba*	
Ribes americanum	*Stellaria longifolia*	
Salix amygdaloides	*Symphyotrichum ciliatum*	
Salix Interior	*Tanacetum vulgare*	
Salsola tragus	*Thalictrum dasycarpum*	
Schoenoplectus pungens	*Thlaspi arvense*	
Schizachyrium scoparium	*Toxicodendron rydbergii*	
Scirpus pallidus	*Triglochin maritima*	
Securigera varia	*Vernonia fasciculata*	
Silphium laciniatum	*Verbena hastata*	
Solidago missouriensis	*Veronica peregrina*	
Solidago mollis	*Viola nephrophylla*	
Sporobolus heterolepis	*Zigadenus elegans*	
Symphyotrichum falcatum	*Zizia aptera*	
Symphyotrichum laeve	*Zizia aurea*	
Tanacetum vulgare		
Thalictrum venulosum		
Thermopsis rhombifolia		
Toxicodendron rydbergii		
Tragopogon dubius		
Trifolium pratense		
Trifolium repens		
Urtica dioica		
Verbena stricta		
Viola nephrophylla		
Viola nuttallii		
Viola pedatifida		
Xanthium strumarium		
Zigadenus elegans		

Seasonal Wetlands Species Retained		
Low Prairie	**Wet Meadow**	**Shallow Marsh**
Achillea millefolium	*Alopecurus arundinaceus*	*Alopecurus aequalis*
Ambrosia psilostachya	*Ambrosia psilostachya*	*Bolboschoenus fluviatilis*
Amorpha canescens	*Anemone canadensis*	*Chenopodium album*
Andropogon gerardii	*Apocynum cannabinum*	*Chenopodium rubrum*
Anemone canadensis	*Artemisia absinthium*	*Cirsium arvense*
Apocynum cannabinum	*Asclepias speciosa*	*Carex atherodes*
Artemisia absinthium	*Bromus inermis*	*Carex laeviconica*
Artemisia ludoviciana	*Calamagrostis canadensis*	*Carex pellita*
Asclepias ovalifolia	*Calamagrostis stricta*	*Eleocharis palustris*
Asclepias speciosa	*Chenopodium album*	*Hordeum jubatum*
Astragalus canadensis	*Cirsium arvense*	*Lemna trisulca*
Bromus inermis	*Convolvulus arvensis*	*Lemna turionifera*
Calamagrostis stricta	*Carex atherodes*	*Lycopus asper*
Cirsium arvense	*Carex brevior*	*Medicago lupulina*
Cirsium flodmanii	*Carex laeviconica*	*Polygonum amphibium var. stipulaceum*
Convolvulus arvensis	*Carex pellita*	*Polygonum lapathifolia L.*
Carex brevior	*Carex sartwellii*	*Phalaris arundinacea*
Carex laeviconica	*Carex vulpinoidea*	*Phragmites australis*
Carex pellita	*Eleocharis palustris*	*Poa palustris*
Elaeagnus commutata	*Elymus repens*	*Potentilla norvegica*
Elymus repens	*Equisetum arvense*	*Rumex crispus*
Equisetum laevigatum	*Equisetum laevigatum*	*Schoenoplectus acutus*
Euphorbia esula	*Euphorbia esula*	*Sonchus arvensis*
Galium boreale	*Glycyrrhiza lepidota*	*Sparganium eurycarpum*
Glycyrrhiza lepidota	*Helianthus maximiliani*	*Spartina pectinata*
Heliopsis helianthoides	*Helianthus nuttallii*	*Teucrium canadense*
Helianthus maximiliani	*Hordeum jubatum*	*Typha x glauca*
Helianthus nuttallii	*Juncus arcticus*	*Utricularia macrorhiza*
Helianthus pauciflorus	*Lycopus americanus*	
Hordeum jubatum	*Lycopus asper*	
Juncus arcticus	*Melilotus officinalis*	
Lactuca tatarica	*Mentha arvensis*	
Medicago lupulina	*Polygonum amphibium var. stipulaceum*	
Medicago sativa	*Polygonum lapathifolia L.*	
Melilotus officinalis	*Phalaris arundinacea*	
Monarda fistulosa	*Poa palustris*	
Polygonum amphibium var. stipulaceum	*Poa pratensis*	
Phalaris arundinacea	*Potentilla anserina*	
Phleum pratense	*Potentilla norvegica*	
Poa pratensis	*Rosa woodsii*	
Potentilla anserina	*Rumex crispus*	

Seasonal Wetlands Species Retained		
Low Prairie	**Wet Meadow**	**Shallow Marsh**
Pediomelum argophyllum	*Salix Interior*	
Rosa arkansana	*Schoenoplectus pungens*	
Rosa woodsii	*Solidago canadensis*	
Rudbeckia hirta	*Solidago gigantea*	
Rumex crispus	*Sonchus arvensis*	
Solidago canadensis	*Spartina pectinata*	
Solidago gigantea	*Stachys pilosa*	
Solidago rigida	*Symphyotrichum ericoides*	
Sonchus arvensis	*Symphyotrichum lanceolatum*	
Sorghastrum nutans	*Symphoricarpos occidentalis*	
Spartina pectinata	*Taraxacum officinale*	
Stachys pilosa	*Teucrium canadense*	
Symphyotrichum ericoides	*Typha x glauca*	
Symphyotrichum lanceolatum	*Urtica dioica*	
Symphoricarpos occidentalis	*Vicia americana*	
Taraxacum officinale		
Teucrium canadense		
Thalictrum dasycarpum		
Thinopyrum intermedium		
Vicia americana		
Zizia aptera		
Zizia aurea		

APPENDIX D. NDRAM FIELD DATA FORM

Metric 1. Buffers and surrounding land use.

1a. Calculate Average Buffer Width

Score	Rating Description
	WIDE. (10 pts)
	MEDIUM. (7 pts)
	NARROW. (4 pts)
	VERY NARROW. (0 pts)
	OTHER (explain)

1b. Intensity of Surrounding Land Use.

Score	Rating Description
	VERY LOW. (10 pts)
	LOW. (7 pts)
	MODERATELY HIGH. (4 pts)
	HIGH. (1 pt)
	OTHER (explain)

	Total for Metric 1 (out of possible 20).

Metric 2. Hydrology, Habitat alteration, and Development.

2a. Substrate/Soil Disturbance.

Score	Rating Description
	NONE. (7 pts).
	RECOVERED. (5 pts).
	RECOVERING. (3 pts).
	RECENT OR NO RECOVERY. (1 pt).
	OTHER (explain)

2b. Plant Community and Habitat Development.

Score	Rating Description
	EXCELLENT. (12 pts)
	VERY GOOD. (10 pts)
	GOOD. (8 pts)
	MODERATELY GOOD. (6 pts)
	FAIR. (4 pts)
	POOR TO FAIR. (2 pts)
	POOR. (0 pts)

2c. Habitat Alteration and Recovery.

Score	Rating Description
	MOST SUITABLE. (10 pts).
	NONE OR NONE APPARENT. (7 pts).
	RECOVERING. (4 pts).
	RECENT OR NO RECOVERY. (1 pt).
	OTHER (explain)

2d. Management.

Score	Rating Description
	Fire or Moderate Grazing. (4 pts)
	Restored, CRP, Hayed, or Idle (2 pts)
	Cropped. (0 pts)
	Other (explain)

2e. Modifications to Natural Hydrologic Regime.

Score	Rating Description
	NONE. (12 pts).
	RECOVERED. (8 pts).
	RECOVERING. (4 pts).
	RECENT OR NO RECOVERY. (1 pt).
	OTHER (explain)

2f. Potential of wetland.

Score	Rating Description
	EXCELLENT. (12 pts).
	GOOD POTENTIAL. (10 pts).
	MODERATE POTENTIAL. (7 pts).
	MODERATELY POOR POTENTIAL. (5 pts).
	POOR POTENTIAL. (2 pts).
	NO POTENTIAL. (0 pts).

	Total for Metric 2 (out of possible 57).

Metric 3. Vegetation

3a. Invasive species.

Score	Rating Description
	ABSENT. (3 pts)
	NEARLY ABSENT. (1 pt)
	SPARSE. (0 pts)
	MODERATE. (-1 pts)
	EXTENSIVE. (-3 pts)

3b. Overall condition

Score	Rating Description
	VERY GOOD (20 pts).
	GOOD (15 pts).
	FAIR (10 pts).
	POOR (5 pts).
	VERY POOR (0 pts).

	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

Score	
	Overall condition rating for wetland (Good, Fair High, Fair Low, or Poor).
Comments:	