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Rate of Vegetation Recovery in Restored Prairie Wetlands

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Abstract

Wetlands are restored to compensate for wetland loss and degradation. To determine the potential rate and success of vegetation recovery in restored wetlands, prairie wetlands of different restoration ages (3 to 23 years since restoration), including drained and natural (embedded within both agricultural and protected landscape), were sampled for vegetation in Alberta, Canada. Vegetation was assessed based on species richness, percentage and cover of hydrophytes, natives and non-natives, and community composition. Analysis of covariance with wetland area as a covariate and non-metric multidimensional scaling results indicated that restored wetlands resembled low-integrity natural wetlands that occurred on agricultural landscapes within 3-5 years of restoration. However, restored wetlands differed in community composition when compared to high-integrity natural wetlands that occurred on protected landscapes. Early establishment of non-native species during recovery, dispersal limitation, and depauperated native seedbank were probable barriers to successful recovery. This differential success of vegetation recovery highlights the need for improved region-specific wetland restoration actions.

Keywords

Wetland, restoration, vegetation, community composition

Co-Authorship Statement

This thesis includes data that will be included in a paper co-authored by S. Salaria, R. Howard, and I. Creed. S. Salaria will be lead author, as she led the implementation of the experimental design, data collection, statistical analysis, and writing of the paper. R. Howard will be co-author, as she contributed to data collection. I. Creed will be co-author, as she supervised the work, directed the experimental design, edited the paper, and provided the financial resources to complete this thesis.

Dedication

I dedicate this thesis to my loving grandparents

Om Prakash Parmar and *Shanti Devi*

Rachchpal S. Salaria and *Ratan Devi*

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List of Abbreviations

ACIMS	Alberta Conservation Information Management System
ANCOVA	Analysis of Covariance
ANOVA	Analysis of Variance
DEM	Digital Elevation Model
DUC	Ducks Unlimited Canada
GIWs	Geographically Isolated Wetlands
ITIS	Integrated Taxonomic Information System
Nat(Agr)	Natural wetlands on Agricultural landscape
Nat(Pr)	Natural wetlands on Protected landscape
NMDS	Non-metric Multidimensional Scaling
PERMANOVA	Permutational Multivariate Analysis of Variance
P-PET	Precipitation - Potential Evapotranspiration
US	United States
VASCAN	Vascular Plants of Canada
WIS	Wetland Indicator Status

1 Introduction

1.1 Problem Statement

Wetlands are among the world's most productive ecosystems (Mitsch & Gosselink, 2007; Kennedy & Mayer, 2002). They provide many ecosystem services to society such as carbon sequestration, water quality improvement, flood control, groundwater recharge, nutrient and biogeochemical cycling, and habitat to a variety of flora and fauna (Marton et al., 2015; Mitsch & Gosselink, 2007; Zedler & Kercher, 2005). Despite this, wetlands have suffered a loss of 54-57% of its area worldwide which continues to take place given pressures from agriculture, urban expansion, industrialization, and resource extraction (Davidson, 2014; Zedler & Kercher, 2005). Canada, which contains one-fourth of the world's wetland area (approximately, 127 million ha), has had an estimated wetland loss of 15.75% between 1800 and late 1980s (Environment Canada, 1991), largely attributed to agricultural intensification (Wiken et al., 2003).

Recently, there has been a shift in public attitude and perception of wetlands as 'wastelands' towards valuing and conserving these ecosystems (Wiken et al., 2003). In response to this, various policies have been adopted at international, national, and provincial scales to mitigate wetland loss and degradation. An important aspect of these policies is to reverse the trend of historical and on-going wetland losses by restoring these ecosystems. Wetland restoration is quite common in the US and increasingly being practiced in Canada as new provincial policies are surfacing, for example, Alberta's Wetland Policy (2013). However, it is not uncommon that a wetland may deviate from its expected recovery path and thus fail to meet goals of structural and functional similarity to natural wetlands (Moreno-Mateos et al., 2017; 2012). Therefore, it is crucial to measure success and failures of wetland restoration to ensure that policy objectives are being met (Wortley et al., 2013). This study specifically assesses wetland restoration in one of the most disturbed regions in Alberta, Canada – the Central Parkland ecoregion within the Prairie Pothole Region. To my knowledge, no other study has yet taken a chronosequence or time-series approach to determine rate and success of vegetation recovery in restored wetlands in this region of Alberta (however see Puchniak (2002) and Wilson et al. (2013)).

1.2 Literature Review

1.2.1 Prairie Wetlands and Their Loss and Degradation

The Prairie Pothole Region is a large physiographic region that stretches over US-Canada (777,000 km²), and contains numerous shallow depressional wetlands often called ‘potholes’ or ‘prairie wetlands’ (Dahl, 2014). Within Canada, the region spans about 386,090 km² covering portions of Alberta, Saskatchewan, and Manitoba (Dahl, 2014). These wetlands were formed by glacial retreat and melt (Wisconsin glaciation) during the Pleistocene Epoch. The region has a strong north-south temperature and east-west precipitation gradient (Johnson et al., 2005), which has largely resulted in wetlands existing along a range of hydrologic conditions (van der Valk, 2005). The prairie wetlands vary from ephemeral, which hold surface water for a very short duration of time after snowmelt and precipitation events, to permanently filled waterbodies (van der Valk, 2005; Stewart & Kantrud, 1971). Due to this variability in water permanence, these wetlands tend to develop concentric zones of vegetation that are characterized by different plant assemblages (van der Valk, 2005; Stewart & Kantrud, 1971). These wetlands are biodiversity hotspots supporting many species at risk and nearly 50% of North America’s waterfowl population (Environment Canada, 2013; Galatowitsch & van der Valk, 1998; Batt et al., 1989).

Despite their importance, prairie wetlands have suffered losses mainly because most of them lack apparent surface water connections to other waterbodies (*aka* geographically isolated wetlands (GIWs)) and therefore they were thought to provide fewer ecosystem services (McLaughlin et al., 2014). These wetlands are rich in nutrient and organic content and provide fertile soils for agriculture use (Kennedy & Mayer, 2002). As such, many wetlands have been subjected to drainage and filling resulting in an estimated loss of nearly 70% within the Canadian prairies (Kennedy & Mayer, 2002), with the settled southern areas experiencing greater wetland losses (Wiken et al., 2003). A detailed account of wetland losses is not possible due to lack or inadequacies of wetland inventory and monitoring programs (Dahl & Watmough, 2007). Generally, small wetlands are more vulnerable to land conversion activities (Watmough & Schmoll, 2007). Bartzen et al. (2010) also concluded that wetlands with lower water permanence are affected the most by agricultural activities and thus are more vulnerable to degradation.

1.2.2 Current Wetland Restoration Efforts

Many policies have been adopted and amended over the years to secure legislative protection to prairie wetlands in recognition of their ecological, economic, and social importance. In the US, the *Clean Water Act* of 1972 under its subsection 404 mandates compensatory measures for wetland loss and damage by restoring, enhancing, and creating wetlands. The intent of the Act is to achieve a “no net loss” of wetland area and thus ecological processes. In Alberta, no such policy existed until 1993 to address wetland mitigation, and the only wetlands protected were on the federal lands by the *Federal Policy on Wetland Conservation* enacted under the *Canadian Environmental Assessment Act* (Rubec & Hanson, 2009). Both, the Alberta Interim Wetland Policy (1993) and the federal policy had a similar intent to achieve “no net loss” of wetland area (Rubec & Hanson, 2009). However, continued wetland loss and degradation in Alberta due to lack of clear guidelines and discrepancies in the interim policy led to the development of Alberta’s Wetland Policy in 2013. Unlike others, this new policy assigns a relative value to a wetland based on its importance to ecological health, water purification, hydrological health and human use, in addition to, its area within the region to ultimately decide and prioritize wetland management and restoration actions (Government of Alberta, 2013).

The Alberta Wetland Policy follows a mitigation hierarchy of avoidance and minimization, and considers wetland compensation as a last resort under which wetland restoration, enhancement, and creation are practiced (Government of Alberta, 2013).

However, avoidance is usually neglected and compensatory measures are often practiced (Clare & Krogman, 2013; Clare et al., 2011). This regulatory approval of wetland loss and degradation in Alberta can only be justified if restored wetlands meet the intended goal of functioning similarly to natural wetlands. While a directive for wetland restoration was issued in 2016 to provide guidance to plan and conduct restoration, a directive to ensure the effectiveness of wetland restoration is yet to be provided (Government of Alberta, 2016). Currently, an Index of Biological Integrity and Floristic Quality Index are being developed in Alberta to monitor wetland health (see Wilson & Bayley, 2012; Rooney & Bayley, 2011) but these require extensive biosurveys with take time and resources. Meanwhile, restorations are on-going hence the urgent need for their assessment.

In the US, research on wetland restoration is extensive and many different biological and physical indicators have been used for the assessment (Wortley et al., 2013). In contrast, research is limited in Canada to guide any restoration actions. Though similar studies can provide knowledge on recovery rate of wetlands, it is still crucial to monitor and evaluate restoration success for a given region owing to its unique physiognomy and biodiversity, inherent variability in recovery rate of wetlands, and differential impact of anthropogenic disturbances (Kentula, 2000).

1.2.3 Theory and Concepts of Wetland Restoration

Ecological theories and concepts related to state-transition models, disturbance, and succession provide a contextual basis to restoration actions. Restoration focuses on bringing an ecosystem either back to its ‘pre-disturbed’ state or a desirable ‘restored’ state (Hobbs, 2007; Hobbs & Harris, 2001). The difference in the two outcomes lies in considering an ecosystem as a static or a dynamic entity (Hobbs, 2007; Hobbs & Harris, 2001). Accordingly, different state-transition models are applied to guide recovery. The models exploit the resilience property of an ecosystem to meet the intended outcomes (Gunderson, 2000). The static view considers that an ecosystem only exists in one stable state and resilience is thus the time taken by an ecosystem to recover to its pre-disturbed state following a disturbance (Gunderson, 2000). The dynamic view considers that an ecosystem can exist in multiple alternative stable states and resilience is a measure of the disturbance which is required to transit an ecosystem to another self-organized stable state that is maintained by a different set of processes and structure (Gunderson, 2000). The degraded state of an ecosystem is itself an alternative state in which new abiotic and biotic conditions are developed, and strong positive feedbacks and interactions among these conditions sometimes provide degraded ecosystem a resiliency to restoration (Suding et al., 2004). Hence, resiliency of such a degraded system needs to be broken to bring a transition to a desirable 'restored' state. Implicit in this model is consideration of complex and different dynamics that exist in alternative states of an ecosystem which may make trajectory to recovery different from trajectory to degradation (Suding et al., 2004). In this manner, the model also acknowledges uncertainty inherent in restoration projects where an existence of multiple trajectories can either cause a successful recovery or a failure (Suding et al., 2004). This makes the use of an alternative state-transition model more valid and acceptable.

A state transition can be brought about by a disturbance which, in this case, is the action taken to restore an ecosystem. The type of action required depends upon the damage to ecosystem, the type of degradative forces acting, and the intended outcome of the restoration (Walker & del Moral, 2008). Also, magnitude, frequency, and duration of restoration action greatly impact recovery of an ecosystem (Walker & del Moral, 2008). Whisenant (1999) suggested two types of thresholds – biotic and abiotic – that a restoration may need to cross to cause an ecosystem transition from degraded to restored state (Figure 1.1). Three likely scenarios exist in this conceptual framework. First, an ecosystem has degraded a little but not crossed any threshold, in which case, it will recover itself (autogenic processes). Second, a biotic threshold has been crossed due to factors like invasion by non-natives or overgrazing, in which case, active restoration actions aimed at removal of non-natives or animal are required to aid recovery. Lastly, an abiotic threshold has been crossed due to factors like impaired hydrology and soil structure, in which case, active restoration actions aimed at restoring physio-chemical structure of ecosystem are required to aid recovery. In this case, there is no point in manipulating biotic factors before restoring abiotic conditions. Abiotic limitations in

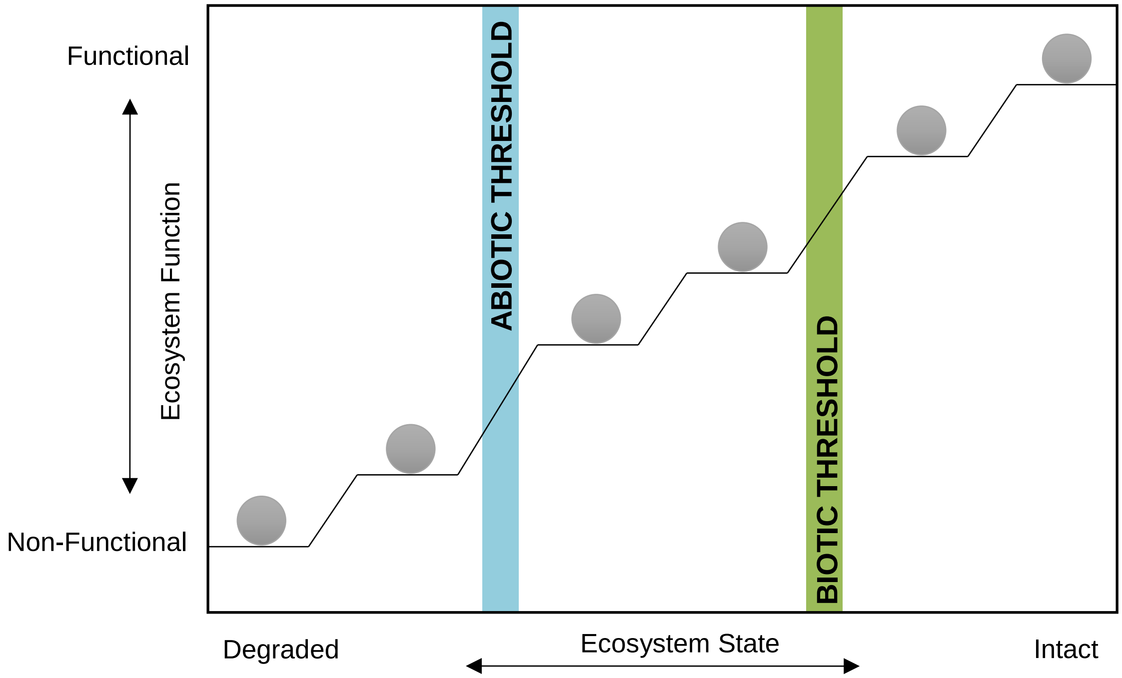

Figure 1.1 State-transition model applied to ecosystem degradation and restoration (adapted from Whisenant, 1999; Hobbs & Harris, 2001). The abiotic and biotic thresholds prevent transition from degraded to restored or intact state.

prairie potholes are typically overcome by restoring natural hydrology of the basin (Galatowitsch et al., 1994).

Disturbance initiates succession on which most restoration efforts rely. In fact, restoration is often considered as ‘a manipulated succession’ to achieve the desired ecosystem state (Young et al., 2001). Succession is a process of sequential and predictable return of vegetation that ultimately progresses toward the development of a climax community (Young et al., 2001; Figure 1.2). In wetlands, this implies an initial colonization by annuals, followed by perennials, and eventually by woody perennials during recovery (Noon, 1996). This colonization and extinction of species depends upon species characteristics, and its interaction with other species and abiotic processes (Young et al., 2001). Due to its simple and deterministic nature, succession theory forms the basis of many restoration policies that aim to achieve similar community composition prior to the degraded state. This theory has, however, been challenged by community assembly theory, which considers existence of rather complex successional trajectories resulting from historical and spatial contingencies leading to development of different community composition than expected (Young et al., 2001). Historical contingency includes variation in the timing of species colonization during recovery (Young et al., 2001). On the other hand, spatial contingency includes constraints posed by attributes like edge and area (Young et al., 2001). To elaborate, presence of dominant species early during recovery may exclude establishment of many other species, especially those which have similar niche requirement (Young et al., 2001). Similarly, proximity to degraded edges causes recruitment of non-native species, and large areas may allow for accelerated succession owing to habitat heterogeneity and within site dispersal opportunities for species (Cook et al., 2005; Young et al., 2001; Young 2000). These contingencies affect successional trajectories and thus community composition of restored ecosystems. A detailed list of site-level morphometric properties that can affect successional trajectories, and hence vegetation community composition of a restored wetland is provided in Table 1.1.

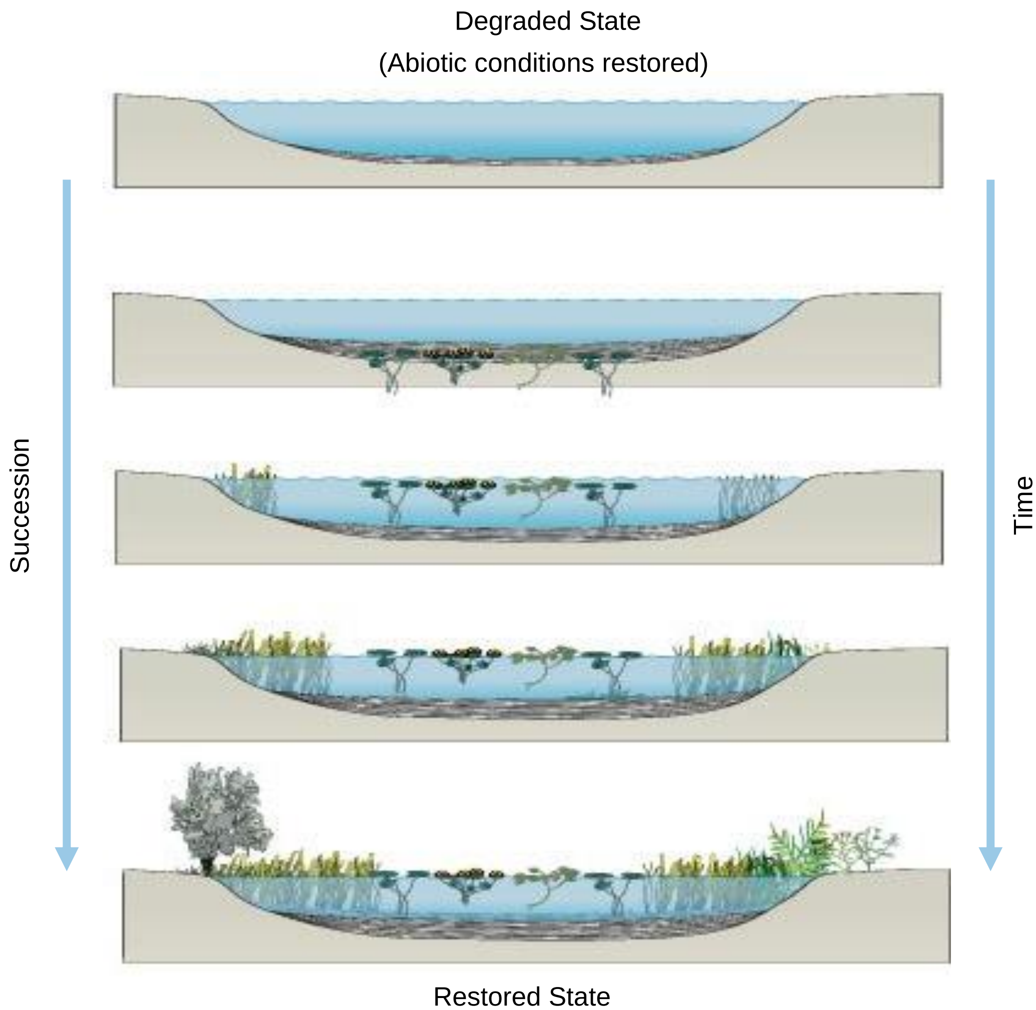

Figure 1.2 Ecological succession in a wetland (Image retrieved and adapted from Mr G's environmental systems, 2009).

Table 1.1 Morphometric properties of wetland that effect vegetation diversity, composition, and distribution.

Morphometric Property	Method	Effect	Reference
Wetland Size	Area	Larger wetlands support higher species diversity due to greater habitat heterogeneity, within site dispersal opportunities, and higher probability of receiving dispersed seeds and propagules.	(Møller & Rørdam, 1985) (Jones et al., 2003) (Mathews et al., 2005) (Rolon & Maltchik, 2006) (Moreno-Mateos et al., 2012) (Kirkman et al., 2012)
Wetland Edge	Edge density, Edge shape, Edge orientation, Edge contrast	Edge influences species richness and distribution pattern. This is dependent on adjacent land-use.	(Ries et al., 2004) (Bowman Cutway & Ehrenfeld, 2010)
Wetland Shape Complexity	P:A ratio, Shape Index = P/ (2√A*π where P is perimeter and A is area	Shape complexity influences species richness. This is dependent on adjacent land-use.	(Moser et al., 2002) (Heegaard et al., 2007)
Wetland Slope	Slope angle, As a measure of soil moisture (for e.g. Topographic Wetness Index)	Slope influences species richness and distribution pattern by its control on soil moisture.	(Collins & Battaglia, 2001) (Moselund et al., 2013) (Forrest, 2010)
Wetland Isolation	Mean distance to nearest wetlands calculated as centroid-centroid, centroid to edge or edge to edge distance, Density within specified area, Isolation Index	Smaller inter-wetland distance facilitates seed dispersal and propagule availability among wetlands leading to higher species diversity.	(Mathews et al., 2005) (Boughton et al., 2010) (Møller & Rørdam, 1985) (Kirkman et al., 2012)
Position in the landscape	Altitude	Species richness decreases with increasing altitude due to temperature differences and restricted habitat availability.	(Rolon, & Maltchik, 2006) (Jones et al., 2003) (Heino, 2002)

1.2.4 Measures of Wetland Recovery and Restoration Success

Various physical, chemical, and biological measures of recovery are used to assess wetland restoration. However, vegetation is the most common measure of recovery because its effect on other biota, links to ecological processes, and sensitivity to disturbances makes it an indicator of ecological integrity of an ecosystem (Ruiz-Jaen & Aide, 2005; Young 2000). Moreover, vegetation sampling is inexpensive and easy to conduct (Ruiz-Jaen & Aide, 2005). A detailed list of vegetation-based metrics is provided in Appendix A that has been used by researchers and resource managers to assess vegetation in an ecosystem. For this study, a suite of commonly used and easily interpreted (to resource managers) vegetation metrics were selected to measure recovery.

Measures of restoration success depends upon the goals of wetland restoration (Hobbs, 2007; Hobbs & Harris, 2001; Kentula, 2000). For example, a wetland may be restored to achieve flood control, sustain populations of certain species, or both, in which case the relative measurement of success may also differ. Most studies have resorted to measuring success (specifically, vegetation recovery) as achieving structural and functional similarity to a set of natural "reference" wetlands (Hobbs, 2007; Ruiz-Jaen & Aide, 2005; Kentula, 2000). Other measures of success include meeting specific requirements of a permit, similarity to replaced natural wetland, similarity to previously restored wetlands, and similarity to natural wetlands prior to the European settlement (Hobbs, 2007; Kentula, 2000). Meeting specific requirements of a permit is often considered to be non-representative of ecological success, whereas similarity to natural wetlands prior to the European settlement is considered an unrealistic measure of success as ecosystems are ever evolving and there is no one fixed desired state to achieve (Hobbs, 2007; Kentula, 2000). On the other hand, information on the specific natural wetland that has been replaced by upland is usually not available which makes it difficult to set it as a base for measuring success (Hobbs, 2007). Similarity to a previously restored wetland, though not practiced, can be useful in cases where natural wetlands are in a comparatively degraded state (Kentula, 2000). More recently, success is also being measured as resilience to anticipated environmental and anthropogenic stress (Kentula, 2000). For this study, restoration success was defined as achieving vegetation community composition that is similar to a set of natural "reference" wetlands (hereafter referred to as natural wetlands), one of the most widely used measures of success.

1.3 Research Objectives and Hypothesis

The purpose of this study is to determine rate and success of vegetation recovery in restored prairie wetlands in the Central Parkland ecoregion of Alberta.

The hypothesis is that rate and success of vegetation recovery in restored wetlands will be a function of wetland morphometrics and age since restoration. The predictions are that: (i) wetland vegetation diversity will increase with larger area, smaller perimeter-to-area ratio, less complex shapes, and gentler slopes, (ii) older restored wetlands (>20 years) will have higher wetland vegetation diversity than younger wetlands (≤5 years), and (iii) restored wetlands will achieve similarity in terms of vegetation community composition to natural wetlands within 10 years of restoration.

The objectives are to: (i) document vegetation diversity and community composition in wetlands, (ii) assess the effect of wetland morphometrics on vegetation diversity and community composition, (iii) assess vegetation diversity and community composition within distinct age classes across a chronosequence of restored wetlands, and (iv) determine success of vegetation recovery in restored wetlands by comparing to nearby natural wetlands of similar size and type.

The significance of this study is its contribution to understanding how restored wetlands perform upon establishment and as they age, and their potential use as a compensatory measure for wetland loss and degradation.

1.4 Thesis Organization

Chapter 1 discusses an urgent need to measure wetland recovery, in addition to, underlying theories and concepts that are needed to understand recovery and measure success. Chapter 2 describes study design, metrics, and statistical analyses used to determine wetland recovery and restoration success. Chapter 3 explains results from the analyses undertaken. Chapter 4 discusses in detail variability in vegetation associated with wetland morphometrics, vegetation recovery across a chronosequence of restored wetlands, and some of the implications for wetland restoration. In doing so, it draws comparison to other similar studies and relates it back to ecological theories and concepts. Chapter 5 presents conclusions, significance of this study, and future research directions.

2 Methods and Materials

2.1 Study Sites

The study sites include 18 restored, 8 natural, and 3 drained prairie wetlands in the Central Parkland ecoregion of Alberta, which covers a small portion of the Canadian Prairies (Figure 2.1). The dominant native vegetation in the region is a mix of aspen and prairie plant communities (Natural Regions Committee, 2006). The landscape mainly comprises glacial till plains, hummocky uplands, and many shallow prairie wetlands formed by the Wisconsin glaciation. Typical soils include Black Chernozemic, Dark Gray Chernozemic, Solonetzic, and Luvisols, in addition to, Gleysolic (humic and orthic) which is a poorly drained soil found especially in wetlands (Natural Regions Committee, 2006). Agricultural intensification and urban development in the region have increasingly placed pressure to convert remnant natural wetlands (Clare et al., 2011; Dahl & Watmough, 2007; Kennedy & Mayer, 2002).

The mean annual temperature is 2.6° C characterized by warm summers and cold winters based on the Canadian Climate Normals for 1981 - 2010 (Environment Canada, 2016). Mean annual precipitation is 446.1 mm, of which 50% falls during June-August (Environment Canada, 2016). The annual water balance is usually negative, with potential evapotranspiration exceeding precipitation (Figure 2.2).

Site selection was based on wetland type (drained, restored, and natural) and wetland class (temporary or seasonal). A wetland inventory was obtained from Ducks Unlimited Canada (DUC, n.d.) and Serran & Creed (2016) to select potential sites for this study. Google Earth imagery was used for preliminary determination of wetland class which was later confirmed during the field visits. Landowner permission and provincial permit were obtained prior to conducting research in the field.

Selected study sites ranged from 0.06 to 1.06 ha. The smaller range of wetland size is due to majority of prairie wetlands within Alberta being typically small (Figure

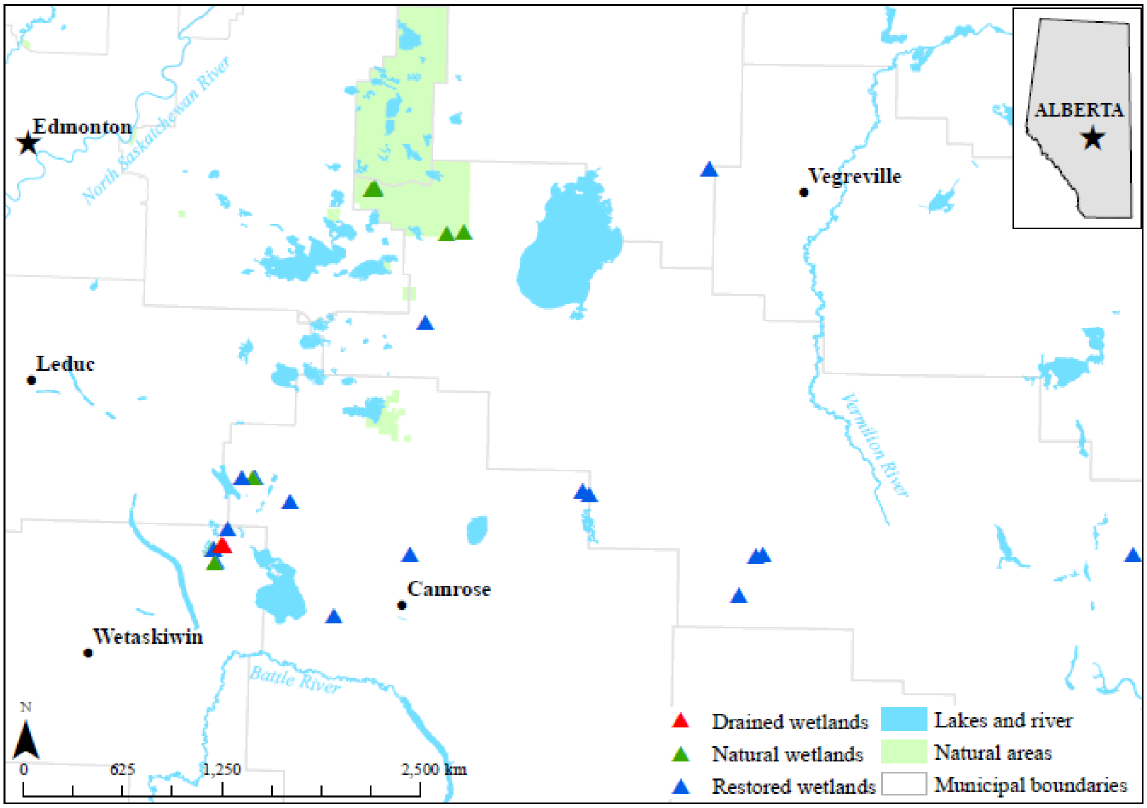

Figure 2.1 Location of study sites which include 18 restored, 8 natural, and 3 drained prairie wetlands in the Central Parkland ecoregion of Alberta, Canada.

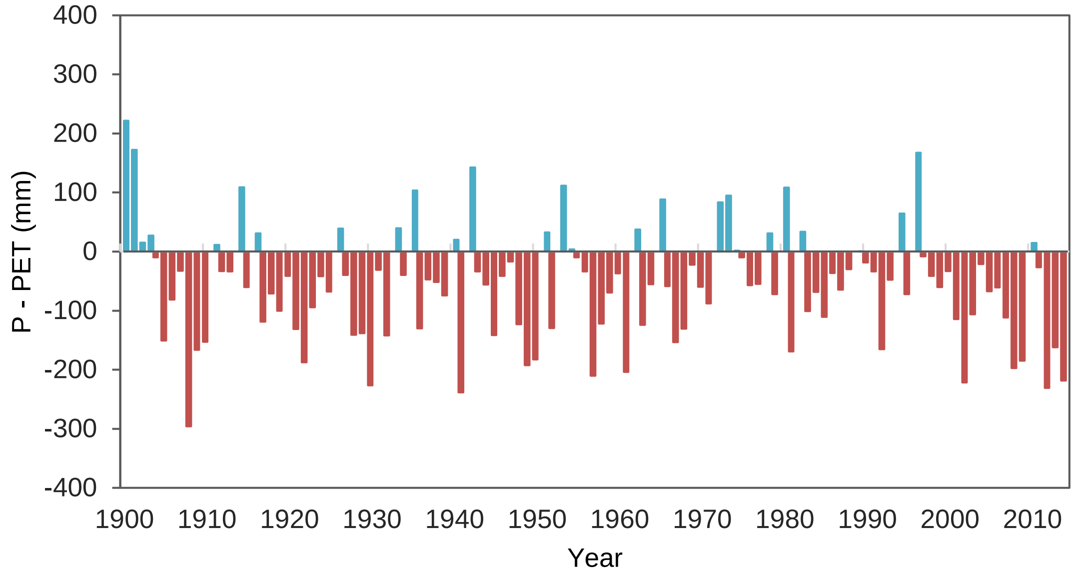

Figure 2.2 Annual precipitation (P) minus potential evapotranspiration (PET) from 1900 - 2010 for Edmonton International Airport. PET was calculated based on Hamon (1961) method.

2.3). Also, restoration efforts within the Parkland ecoregion are highly skewed towards smaller wetlands. Of the 770 wetlands restored between 1957 - 2015, 63.12% are ≤ 1 ha and 83.90% are ≤ 5 ha (DUC, n.d.). Most wetlands are restored by placing earth berms on drainage ditches to restore hydrology of the basin while few are restored by using an engineered structure such as a rock weir to handle large volumes of water during snowmelt and precipitation events.

Selected restored wetlands were on agricultural landscape and aged 3-23 years at the time of sampling in 2016. They were restored by DUC by constructing earth berms on drainage ditches to restore hydrology of the basin. Though these wetlands were left for subsequent natural re-colonization, the earth berms were often seeded at 30 lbs per acre with an equal portion of grass seed mix (usually containing *Bromus riparius* Rehm., *Medicago sativa*, *Schedonorus arundinaceus*, and *Elymus trachycaulus*) and *Hordeum vulgare* to provide berms with stability during flooding events, and to suppress growth of weeds by competing for nutrients (R. Hunka, personal communication, January 3, 2017). Weeds such as *Cirsium arvense* and *Thalspi arvense* were controlled by spraying Roundup. The uplands surrounding restored sites were usually grazed, hayed, or left idle on a rotation like basis. This difference in management practice could not be considered in this study as grazing/ haying happened after the field sampling.

Of the eight natural wetlands, three were selected on the agricultural landscape (Nat(Agr)) and five on the protected landscape (Nat(Pr)) to capture dynamics of natural wetlands within the region. Nat(Agr) represents low-integrity natural wetlands whereas Nat(Pr) represents high-integrity natural wetlands. Selection of several natural wetlands ensures a robust assessment of recovery as it takes into consideration that a restored wetland may undergo different paths of recovery (Ruiz-Jaen & Aide, 2005; Kentula, 2000). Additionally, drained sites were also selected to assess wetland conditions prior to restoration. Detailed site description is provided in Table 2.1. A drained, restored, and natural wetland is shown in Figure 2.4.

(a)

(b)

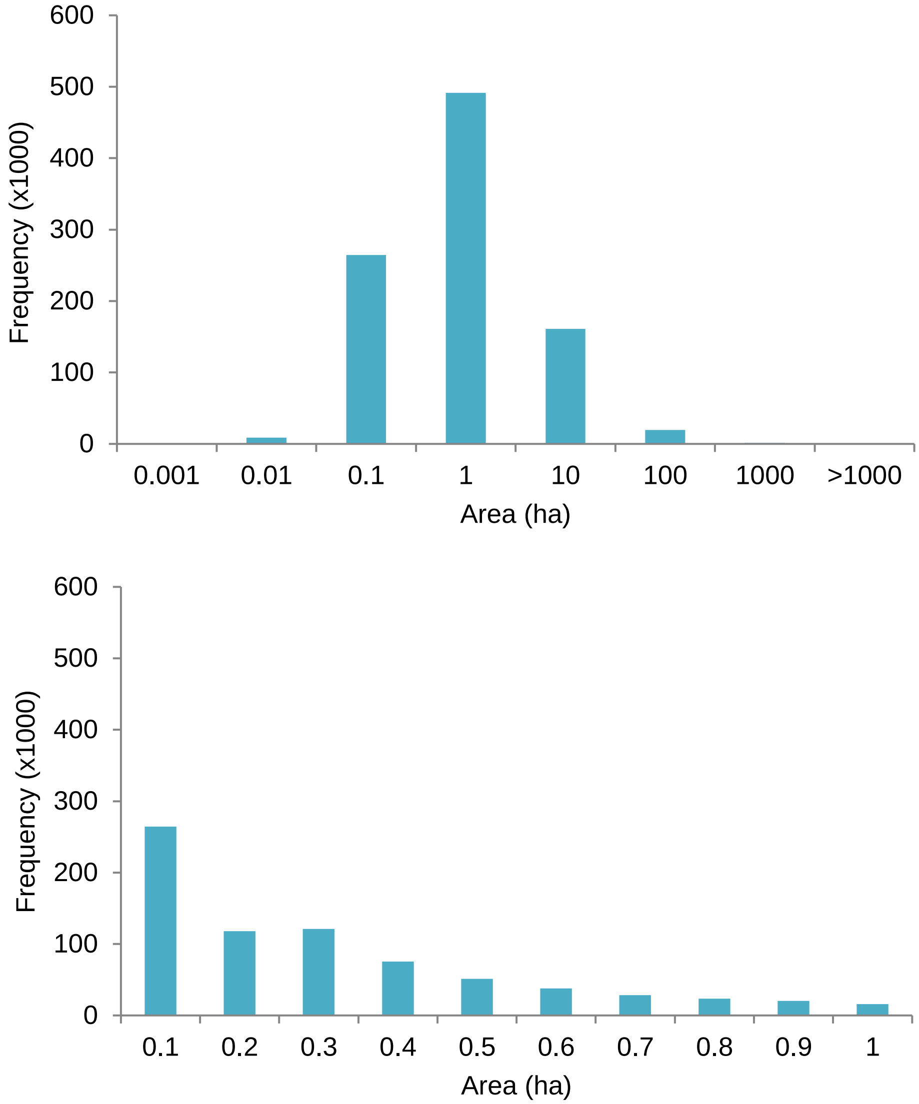

Figure 2.3 Frequency distributions of (a) the area of all the wetlands in the Prairie Pothole Region of Alberta (b) the area of wetlands ranging from 0.1-1 ha in the Prairie Pothole Region of Alberta from the Canadian Wetland Inventory (DUC, 2016).

Table 2.1 Description of study sites located in the Central Parkland ecoregion of Alberta, Canada. Agr stands for agriculture landscape and Pr stands for protected landscape.

Site ID	Age (2016)	Class	Type	Area (ha)
CUR1	0	-	Drained	0.067
CUR2	0	-	Drained	0.069
CUR3	0	-	Drained	0.223
FOR1	3	Seasonal	Restored	0.298
ROP1	3	Seasonal	Restored	0.235
ABB1	4	Seasonal	Restored	0.288
BOW1	4	Seasonal	Restored	0.282
OZM1	5	Temporary	Restored	0.123
LAB1	6	Seasonal	Restored	0.248
BUS1	7	Seasonal	Restored	0.075
NAS1	8	Seasonal	Restored	0.107
HEN1	9	Temporary	Restored	0.241
REU1	9	Temporary	Restored	0.059
BOW2	11	Temporary	Restored	0.089
BOW3	11	Temporary	Restored	0.155
FER1	14	Seasonal	Restored	0.082
FER2	14	Seasonal	Restored	0.531
MCN1	14	Seasonal	Restored	1.062
RAU1	21	Seasonal	Restored	0.998
MIT1	22	Seasonal	Restored	0.158
AMB1	23	Temporary	Restored	0.191
CLBCD1	-	Seasonal	Natural (Pr)	0.459
CLBID8	-	Temporary	Natural (Pr)	0.883
CLBIM1	-	Seasonal	Natural (Pr)	0.974
CLBRD2	-	Temporary	Natural (Pr)	0.584
CLBRD3	-	Seasonal	Natural (Pr)	0.107
INT1	-	Seasonal	Natural (Agr)	0.837
INT2	-	Seasonal	Natural (Agr)	0.799
INT3	-	Seasonal	Natural (Agr)	0.737

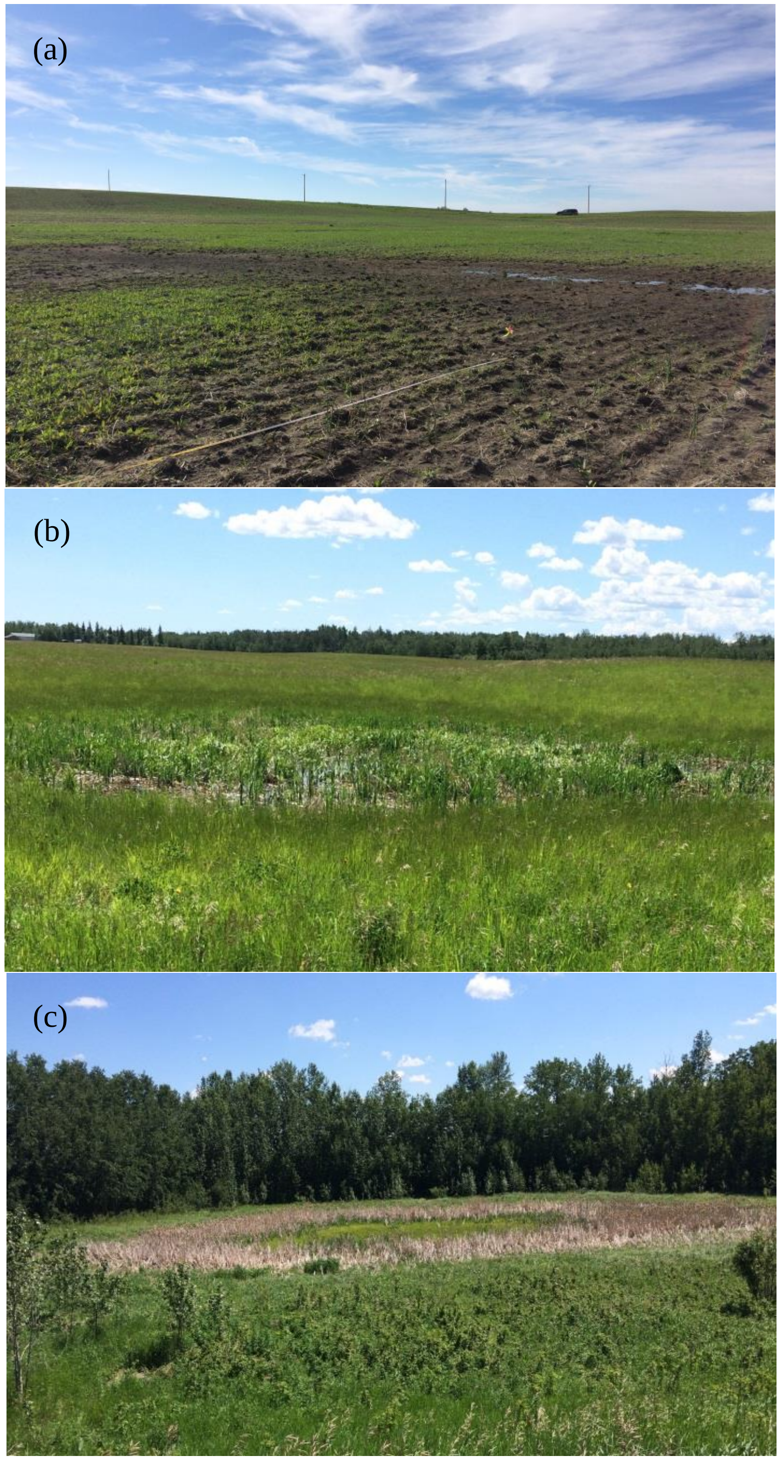

Figure 2.4 Images of (a) drained, (b) restored, and (c) natural prairie wetlands in the Central Parkland ecoregion of Alberta, Canada.

2.2 Selection of Age Classes

A chronosequence approach (space-in-time) was used to determine rate and success of vegetation recovery in restored wetlands. Age classes 0 (n = 3), 3-5 (n = 5), 6-10 (n = 5), 11-15 (n = 5), and >20 (n = 3) were selected. An age class of 0 represents drained sites, as these mark initial conditions of a wetland undergoing recovery. Recovery rates and trajectories of vegetation-based metrics may differ (Mathews et al., 2009), and therefore a longer time scale of >20 allowed measurement of potential differences among different age classes as well as wetland restoration success.

2.3 Wetland Delineation

Wetland boundaries were confirmed in the field based on inspection of vegetation and soil characteristics at regular intervals (25-50 m) along the wetland boundary. Dominance of wetland plant communities, and presence of hydric soil characteristics, such as thick organic layer, redoximorphic features like gleying/mottling within 30 cm of soil, and or oxidized rhizospheres were used to verify the wetland boundary (Government of Alberta, 2015). Field verified wetland boundaries were then digitized using the editor tool in ArcGIS version 10.4 (ESRI, Redlands, CA). Digitized boundaries were used to calculate wetland area (ha) and perimeter-to-area ratio (m^{-1}) using the geometry function. Shape Index (McGarigal & Marks, 1995) was calculated as,

$$SI = \frac{P}{2\sqrt{\pi A}}$$

where P is perimeter (m) and A is area of the wetland (m^2). SI measures the departure of a shape from circle such that a wetland with irregular boundaries has SI value greater than 1. Being dimensionless, this index allows comparisons to be drawn among wetlands of different sizes. Slope (percent rise) was calculated from the province wide 25 m^2 hydrologically corrected DEM (Alberta Environment and Parks, 2008) because a finer resolution DEM was unavailable for the region. The resolution of the DEM was changed to 5 m^2 using a nearest neighbor resampling method to calculate slope for relatively smaller wetlands. The nearest neighbor resampling is an interpolation method that allows to retain original cell value with a maximum spatial error equal to half the cell size (ESRI, 2017). A mode of slope was finally taken using the zonal statistics as a table function.

2.4 Vegetation Sampling

Wetlands were classified based on the Stewart and Kantrud Classification System (1971). This system classifies wetlands based on vegetation and water permanence. Eight sites were classified as temporary wetlands (Class II) and 18 sites were classified as seasonal wetlands (Class III). Temporary wetlands have a central wet meadow zone and usually hold water for only a few weeks after snowmelt and precipitation events. Seasonal wetlands have a central emergent zone, in addition to, outer wet meadow zone and usually hold water till mid-summer.

Vegetation was sampled in each wetland once during the summer period from June to August, 2016. Summer corresponds to the peak growing season in the region. A stratified random sampling design was used to capture vegetation heterogeneity across the hydrologic gradient of the wetland as represented by different vegetation zones (Little, 2013) (Figure 2.5). The first transect was placed starting at the deepest point near the centre of the wetland and moving towards the boundary. Subsequently, three additional transects were placed for a total of 4 transects per wetland, placed approximately 90° apart. This method ensured a good coverage of the wetland vegetation. A series of quadrats were then put randomly along transects to collect replicate samples in each vegetation zone. 1 m^2 quadrat was used to sample herbaceous vegetation and vegetation <1 m in height, a 25 m^2 quadrat was used to sample shrubby/woody vegetation (>1 m) (as required), and a 100 m^2 quadrat was used to sample trees (as required). In cases where vegetation zones were small, quadrats were moved slightly off the transect to collect non-overlapping replicate samples of vegetation. The total number of quadrats sampled varied among sites due to presence of different number of vegetation zones in each wetland class.

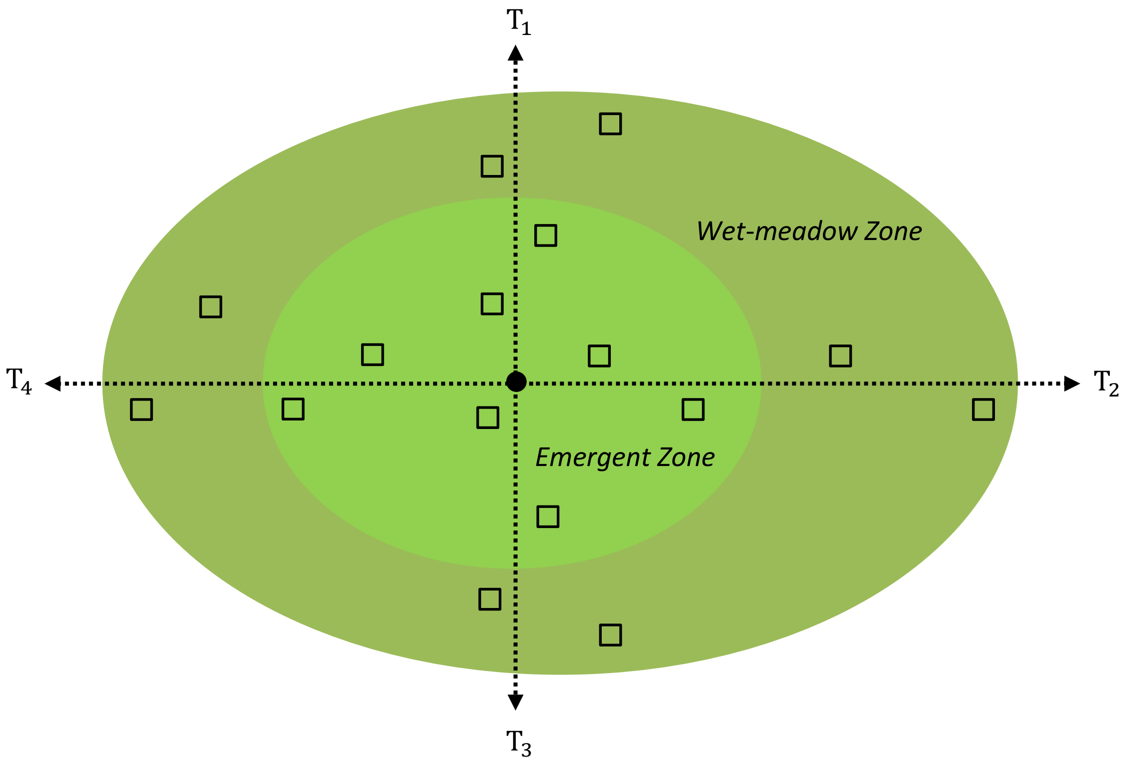

Figure 2.5 A stratified random sampling design to capture vegetation heterogeneity across the hydrologic gradient of the wetland as represented by different vegetation zones. T 1 - 4 represent transects and square boxes represent quadrats.

Several guides such as Tannas (2001, 2003, 2004), Lahrig (2003), Harris & Harris (2001), Bubbar et al. (2000), Johnson et al. (1995), and Moss (1983) were used to identify plants. Most plants were identified to species level while some could only be identified to genus level. Nomenclature closely followed the Integrated Taxonomic Information System (ITIS, https://www.itis.gov), a database which provides reliable taxonomic information on North American flora by adhering to the standards set by International Code of Botanical Nomenclature. Additionally, Database of Vascular Plants of Canada (VASCAN, http://data.canadensys.net/vascan/search) was consulted for a few species whose name could not be identified in ITIS. Unknown species were collected, dried, and stored in a plant press to be later identified at the Western University Herbarium. If a species could not be identified at all, an original name was given and distinguishable plant traits were noted. This helped to keep track of the unidentified species when found in other sites.

Species presence and percent-cover were noted within each quadrat. An 8-point cover classification system <1 %, 1-5 %, 6-10 %, 11-25 %, 26-33 %, 34-50 %, 51-75 % and >75 % was used to estimate percent-cover (Mueller-Dumbois & Ellenberg, 1974), and to minimize any observer bias (Little, 2013). A mid-point of these cover classes was then used and averaged to calculate the percent-cover of each species. Additionally, a random walk through known as Relevé technique (Mueller-Dumbois & Ellenberg, 1974) was conducted for about 30 min (+/- 10 depending upon area) within the wetland to record any rare species, species occurring in patches, or species not previously identified through quadrat sampling. This helped in compiling a comprehensive list of plant species for each wetland, which included species identified through both quadrat sampling and relevé walk.

2.5 Calculation of Vegetation-based Metrics

Vegetation diversity metrics were calculated for each wetland using a list of plant species identified at site scale (comprehensive plant list) and 1 m^2 quadrat scale. Species richness was measured as a count of different species, and cover of different plant guilds such as hydrophytic species, native species, and non-native species was measured as average percent-cover of species in each wetland. Community composition was measured only at site scale to detect differences among wetlands.

2.5.1 Species Richness

Species richness for each wetland was calculated as a total number of observed species at site scale. However, this metric was biased due to its inherent dependence on sampling intensity. Therefore, species richness was also estimated from 1 m^2 quadrat data based on species accumulation curves that can be rarefied to a smaller sample size or extrapolated to a larger sample size to make meaningful comparisons.

Species accumulation curves for each wetland were constructed based on method by Colwell et al. (2012) to estimate species richness from a pooled set of quadrats. This method assumes that even after adequate sampling has been achieved, some species remain undetected. Hence, an asymptotic species richness estimator is used which calculates undetected species and gives an estimate of true species richness at a given level of sampling effort. Chao2 is a recommended estimator for species presence-absence data (Colwell et al., 2012) and works much better than simply fitting a mathematical curve to the data (Hortal et al., 2006). Species richness is estimated as,

$$S_{chao2} = S_{obs} + \frac{(t-1)}{t}\frac{Q_1^2}{2Q_2} \quad \text{when } \mathrm{Q}_2 > 0$$

$$S_{chao2} = S_{obs} + \frac{(t-1)}{t}\frac{Q_1(Q_1-1)}{2(Q_2+1)} \quad \text{when } \mathrm{Q}_2 = 0$$

where S_{obs} is observed species, t is number of quadrats, Q_1 is number of species that occur only once, and Q_2 is number of species that occur only twice (Colwell et al., 2012). The analysis was conducted using EstimateS version 9.1 (Colwell, 2013).

2.5.2 Hydrophytic Species

Hydrophytes are species that are typically found in wetlands as compared to uplands. This includes obligate, facultative wetland, and facultative species (Lichvar et al., 2012). Obligate species are always found in wetlands, facultative wetland species are usually found in wetlands, and facultative species are found both in wetlands and uplands (Lichvar et al., 2012). Each species was assigned its Wetland Indicator Status (WIS) based on the National Wetland Plant List (U.S. Army Corps of Engineers, 2016; Lichvar et al., 2016). Plants identified to genus level were assigned a status by considering all species within that genus that had their respective ranges in my study area and for which the status was known. Flora of Alberta (Moss, 1983) was used for this purpose. 52 species whose status could not be identified were excluded from the analysis. Both percentage of hydrophytic species at site scale and percent-cover of hydrophytes at quadrat scale were calculated.

2.5.3 Native and Non-native Species

Each species was assigned its nativity status based on ACIMS List of Vascular Plants (2015). Plants identified to genus level were assigned status by considering all the species within that genus that had their respective ranges in my study area and for which the status was known. Flora of Alberta (Moss, 1983) was used for this purpose. 32 species whose status could not be identified were excluded from the analysis. Both percentage of native and non-native species at site scale, and percent-cover of natives and non-natives at quadrat scale were calculated.

2.5.4 Sensitive Species

Sensitive species included plants with relatively small distributional ranges, small population sizes, and occurrences of $\leq$100 in Alberta which makes them vulnerable to extirpation especially because of anthropogenic disturbances. This corresponds to the sub-national conservation status rank of S1-S3 as identified in ACIMS (2015).

2.5.5 Community Composition

Similarity in community composition was determined based on species presence-absence data and by Sørensen Index (Sørensen, 1948) which is given as,

$$SI = \frac{2a}{2a + b + c}$$

where SI is Sørensen Index, a is number of shared species between two sites, and b and c are number of species present only in one of the sites. Sensitive species identified above based on ACIMS (2015) were excluded from the analysis to remove any unnecessary variability in the data (McCune & Grace, 2002).

2.6 Statistical Analysis

The assumptions of normality and equal variance were tested using *Shapiro Wilk test* and *Levene's test* to select appropriate statisitical tests - parametric or non-parametric. A spearman rank correlation was used to assess the relationship between vegetation-based metrics (except community composition) and wetland morphometrics. The correlation was also determined between different wetland morphometrics to test for multicollinearity and select suitable covariates for analysis of covariance (ANCOVA). The assumptions of normality and equal variance between groups were checked, in addition to, homogeneity of regression slopes to conduct ANCOVA to detect if any statistically significant differences in vegetation-based metrics (except community composition) exist among wetlands of different age classes, Nat(Agr), and Nat(Pr). The observed species richness was square-root transformed to meet the assumptions of normality. Pairwise comparisons were conducted using *Sidak test*. All statistical tests were performed in SPSS version 24 (IBM Corp, Armonk, NY) at a significance level of 0.05.

A non-metric multidimensional scaling (NMDS) was performed on Sørensen Index to analyze similarities in community composition among sites. NMDS is a method recommended for analyzing community composition among sites because, unlike other ordination techniques, it does not assume linear relationships, makes few assumptions about the dataset, and can be performed on any similarity measure (McCune & Grace, 2002; Clarke, 1993). It also attempts to closely preserve the rank order of similarities in a low dimensional species space such that sites with similar community composition are plotted closer than others (McCune & Grace, 2002; Clarke, 1993). It should be noted that direction, orientation, and scaling of axes in NMDS is arbitrary (Oksanen et al., 2017). NMDS was run 150 times to select the best possible solution with a recommended stress (goodness of fit) of below 0.2 (McCune & Grace, 2002; Clarke, 1993). Wetland area, perimeter-to-area ratio, shape index, slope, and age classes (including Nat(Agr) and Nat(Pr)) were fitted on the ordination to determine correlation between community structure and wetland morphometrics, and if restored wetlands achieved similarity in terms of vegetation community composition to natural wetlands. Morphometric variables were standardized before running the ordination to have a mean of 0 and standard deviation of 1. NMDS analysis and variable fitting was performed in R using *metaMDS()* and *envfit()* functions in vegan package (Oksanen et al., 2017; RStudio, Boston, MA). Significant difference in community composition among sites of different age classes, Nat(Agr), and Nat(Pr) was tested using PERMANOVA (method = unrestricted permutation of raw data, permutation = 9999) in Primer version 7 (Clark & Gorley, 2015). This test has the advantage of handling an unbalanced study design and testing for significant differences with an approach similar to ANOVA. The homogeneity assumption of PERMANOVA was confirmed by running *PERMDISP* test.

3 Results

3.1 Wetland Delineation

Observational records of vegetation, soil, and hydrology that were used to either extend or truncate the desktop delineated boundary of each wetland is summarized in Appendix B. Mean wetland area was 0.38 ha (SD = ±0.33), mean perimeter-to-area ratio was 0.10 m^{-1} (SD = ±0.04), mean shape index was 1.26 (SD = ±0.33), and mean slope was 1.78 percent rise (SD = ±1.62). Spearman rank correlation results revealed that area was negatively correlated to perimeter-to-area ratio ($r = -0.911$, $p < 0.00001$), and slope ($r = -0.455$, $p = 0.013$). There was a positive correlation between perimeter-to-area ratio and slope ($r = 0.582$, $p < 0.0001$). In contrast, shape index was not correlated to any other wetland morphometrics.

3.2 Vegetation Sampling

A total of 188 plant species were identified across 40 families, of which up to 29 species remained unknown (i.e., they could not be identified at their genus level, Appendix C). Dominant families included Poaceae (35 species), Asteraceae (23 species), Cyperaceae (13 species), and Rosaceae (12 species). Native species constituted 66.48%, non-natives 16.48%, and hydrophytes 50% of the total species identified. *Alopecurus pratensis*, *Plantago major*, and *Sonchus arvensis* were the only hydrophytes that were non-native species. Species present in at least 75% of sites were *Agropyron sp.*, *Bromus inermis*, *Carex atherodes*, *Cirsium arvense*, *Mentha arvensis*, *Poa palustris*, *Rumex occidentalis*, *Salix petiolaris*, *Sonchus arvensis*, and *Taraxacum officinale*.

3.3 Variability in Vegetation Associated with Wetland Morphometrics

Wetland area, perimeter-to-area ratio, and slope were associated with species diversity to varying degrees (Table 3.1). In general, wetland area was associated with

Table 3.1 Spearman rank correlation values between wetland morphometrics and vegetation diversity metrics. Significant correlations are bolded (p values are given in brackets, $\alpha = 0.05$).

Metrics	**Area (ha)**	**Perimeter-to-area Ratio (m^{-1})**	**Shape Index**	**Slope (% rise)**
Observed Species Richness	**0.725 (<0.00001)**	**-0.640 (0.0001)**	0.132 (0.493)	-0.195 (0.308)
% Hydrophytes	0.242 (0.203)	-0.148 (0.439)	0.282 (0.136)	-0.133 (0.488)
% Natives	**0.551 (0.002)**	**-0.492 (0.006)**	0.072 (0.706)	**-0.411 (0.027)**
% Non-natives	**-0.477 (0.009)**	**0.447 (0.015)**	0.038 (0.843)	**0.483 (0.008)**
Estimated Species Richness	**0.474 (0.009)**	-0.335 (0.0751)	0.190 (0.321)	0.094 (0.624)
Percent-cover of Hydrophytes	**0.414 (0.025)**	-0.258 (0.175)	0.349 (0.063)	-0.014 (0.942)
Percent-cover of Natives	**0.429 (0.020)**	-0.272 (0.151)	0.345 (0.066)	-0.069 (0.720)
Percent-cover of Non-natives	0.099 (0.608)	0.001 (0.995)	0.054 (0.780)	**0.456 (0.013)**

most of the vegetation diversity metrics. It was positively correlated with species richness at both site and quadrat scale. In addition to this, it was also positively correlated with percentage and percent-cover of natives, and percent-cover of hydrophytes. Wetland area was however negatively correlated with percentage of non-native species. Perimeter-to-area ratio was negatively associated with species richness at site scale. It was negatively correlated with percentage of natives but positively correlated with percentage of non-native species. Similarly, steeper slope was negatively correlated with percentage of natives but positively correlated with percentage and percent-cover of non-native species. In contrast, wetland shape had no significant association with species diversity.

A final two-dimensional NMDS solution was selected to display the vegetation community composition of drained, restored, and natural wetlands. The iterative algorithm of NMDS stopped after 20 random starts when it reached a similar minimum stress twice. A solution with a stress of 0.15 was thus accepted. The correlation-like statistics, which measures the goodness of fit of the NMDS, had a value of 0.97 for 'non-metric fit' and 0.94 for 'metric fit' (Appendix D).

Morphometric variables fitted onto the NMDS ordination using *envfit()* function in R revealed wetland area ($r^2 = 0.287$, $p = 0.013$) and perimeter-to-area ratio ($r^2 = 0.248$, $p = 0.022$) to be significant (but weakly so) in explaining some dissimilarity in vegetation community composition among sites (Figure 3.1). The direction of fitted variables indicated larger areas associated with natural wetlands and higher perimeter-to-area ratio of few drained and restored wetlands.

3.4 Recovery of Vegetation Across a Restoration Chronosequence

The rate and success of vegetation recovery in restored wetlands is described below. Only wetland area was selected as a covariate because of its association with most vegetation diversity metrics and to avoid statistical redundancy caused by correlated covariates.

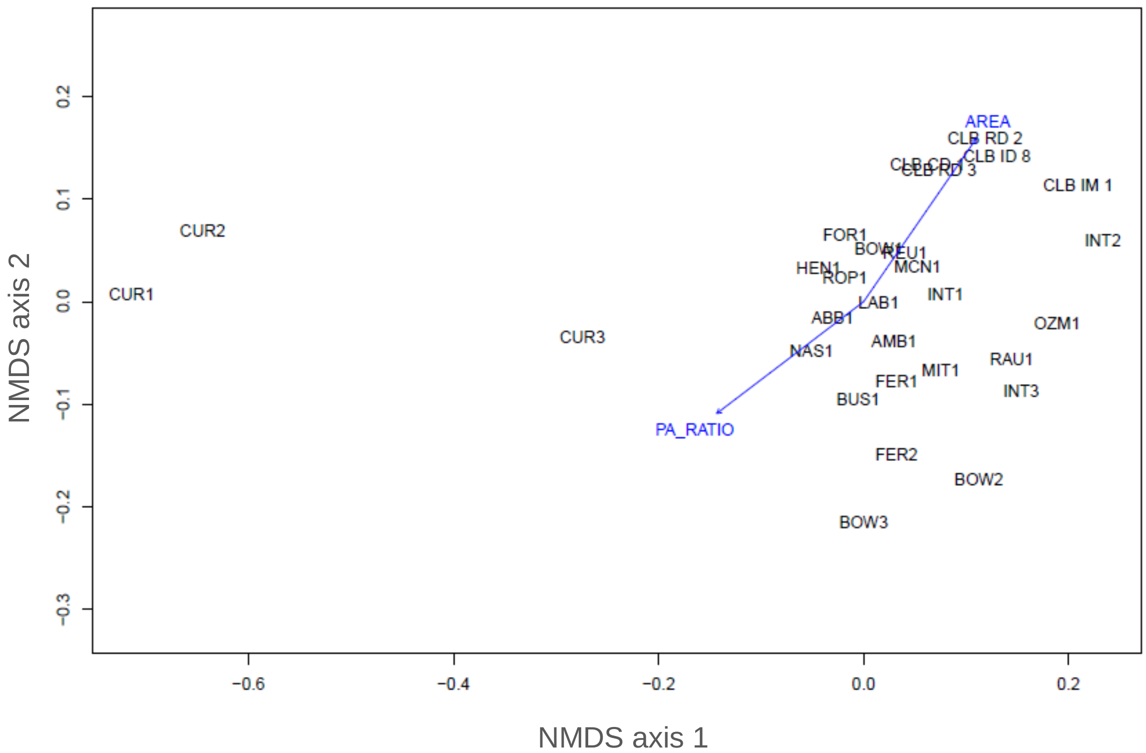

Figure 3.1 Wetland morphometrics fitted on the NMDS ordination of community composition. Only morphometrics (area and perimeter-to-area ratio) that were significantly correlated to ordination of community composition are shown. The direction of arrow represents change in morphometry, and its relative length represents correlation between morphometrics and ordination (Oksanen et al., 2017).

3.4.1 Species Richness

Mean observed species richness was 39.79 species per site (SD = ±14.85). The youngest (3-5 years) and oldest (> 20 years) restored age classes had a higher mean observed species richness than other age classes (15.00 (SD = ±7.21), 45.60 (SD = ±6.50), 33.40 (SD = ±4.93), 33.00 (SD = ±15.68), and 39.00 (SD = ±8.00) for age classes 0, 3-5, 6-10, 11-15, and >20), but comparatively lower than Nat(Agr) (49.67, SD = ±14.15) and Nat(Pr) (56.60, SD = ±5.81). ANCOVA results confirmed a statistically significant difference in observed species richness among age classes and natural wetlands ($F_{(6,21)} = 8.851$, $p < 0.0001$, partial $\eta^2 = 0.717$, observed power = 0.999). Pairwise comparisons revealed that for observed species richness, the drained class had a significantly lower species richness than others except age class 11-15 years (age classes 3-5 ($p < 0.0001$), 6-10 ($p = 0.005$), 11-15 (0.128), >20 ($p = 0.034$), Nat(Agr) ($p = 0.045$), Nat(Pr) ($p < 0.0001$)), and the age class 11-15 years had a significantly lower species richness than the youngest (3-5 years) restored age class ($p = 0.039$) and Nat(Pr) ($p = 0.20$) (Figure 3.2a).

Mean estimated species richness at the same level of sampling effort was 26.86 species per site (SD = ±10.85). The youngest (3-5 years) restored age class still supported a higher mean estimated species richness of 28.60 species per site (SD = ±3.71) comparable to that of Nat(Agr) (28.33, SD = ±6.03). However, species richness was low compared to Nat(Pr) which had a mean of 44 species per site (SD = ±5.34). ANCOVA results confirmed a statistically significant difference in estimated species richness among age classes and natural wetlands ($F_{(6,21)} = 8.386$, $p < 0.0001$, partial $\eta^2 = 0.706$, observed power = 0.999). Pairwise comparisons revealed that for estimated species richness, the drained class had a significantly lower species richness than both the youngest (3-5 years) restored age class ($p = 0.049$), and Nat(Pr) ($p < 0.0001$). In addition, restored age classes 6-10, 11-15, >20, and Nat(Agr) had a significantly lower species richness than Nat(Pr) (age classes 6-10 ($p = 0.045$), 11-15 ($p = 0.001$), >20 ($p = 0.003$), and Nat(Agr) ($p = 0.014$)) (Figure 3.2b).

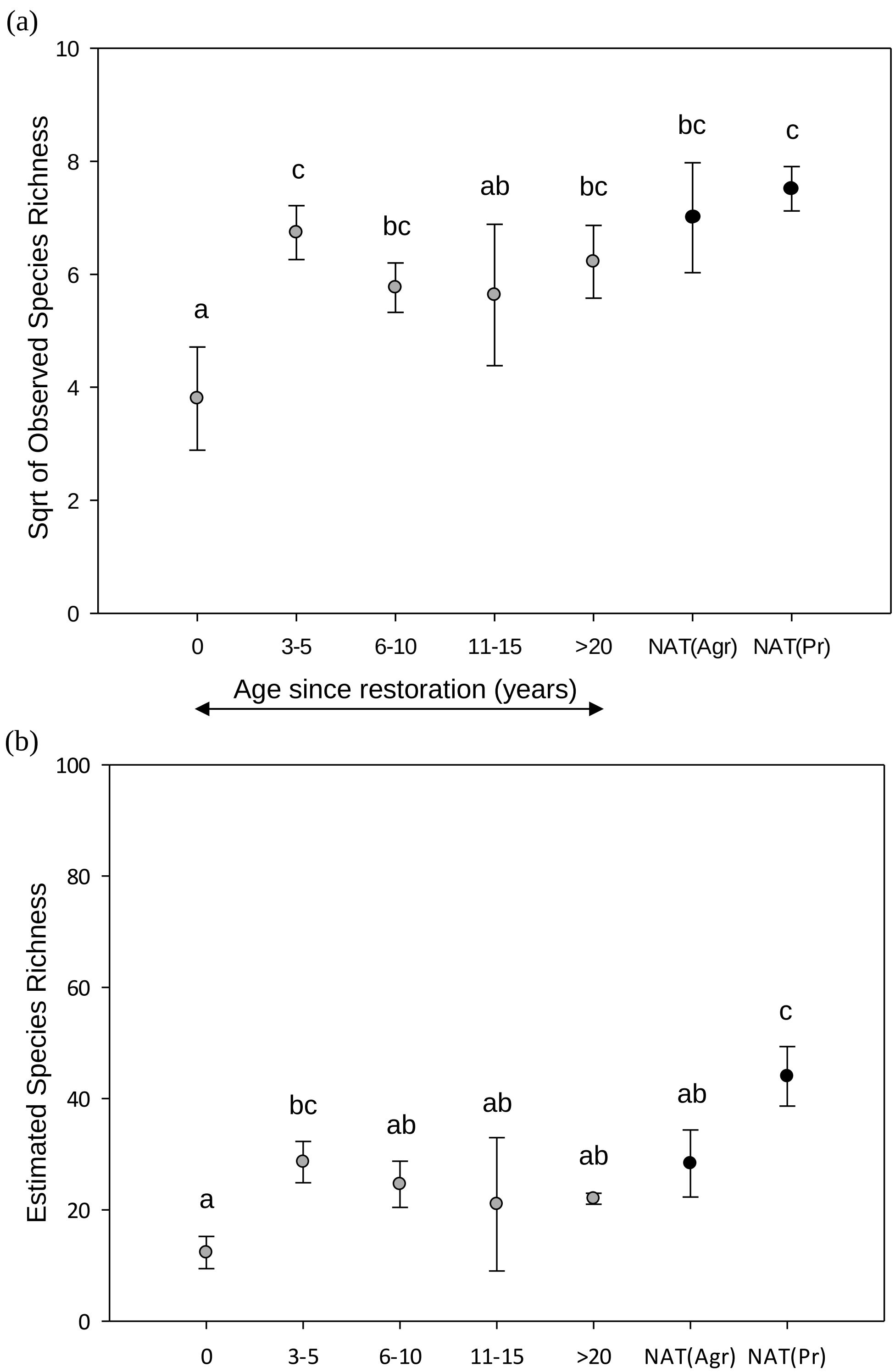

Figure 3.2 Mean ± SD (a) observed species richness, and (b) estimated species richness across a chronosequence of restored wetlands. Age 0 represents drained wetlands. Natural wetlands are represented by black circles. Letters indicate significant differences.

3.4.2 Hydrophytic Species

Mean percentage of hydrophytes was 54.63 (SD = ±13.64). The drained class had a lower mean percentage of hydrophytic species (19.31, SD = ±17.25) in comparison to restored age classes 3-5 (59.98, SD = ±4.71), 6-10 (59.24, SD = ±5.92), 11-15 (57.89, SD = ±3.03), >20 (54.81, SD = ±1.61), Nat(Agr) (56.16, SD = ±3.31), and Nat(Pr) (61.56, SD = ±2.86) (Figure 3.3a). ANCOVA was not conducted on percentage of hydrophytic species because of the significant interaction effect by area ($F_{(6,15)} = 15.063$, $p < 0.0001$, partial $\eta^2 = 0.858$, observed power = 1).

Mean percent-cover of hydrophytes was 39.39 (SD = ±21.26). The drained class had a lower mean percent-cover of hydrophytic species (6.11, SD = ±10.51) in comparison to restored age classes 3-5 (41.40, SD = ±11.76), 6-10 (51.16, SD = ±18.18), 11-15 (33.83, SD = ±23.68), >20 (36.71, SD = ±22.43), Nat(Agr) (38.31, SD = ±26.03), and Nat(Pr) (53.39, SD = ±13.74). ANCOVA results confirmed a statistically significant difference in percent-cover of hydrophytic species among age classes and natural wetlands ($F_{(6,21)} = 2.741$, $p = 0.040$, partial $\eta^2 = 0.439$, observed power = 0.753). Pairwise comparisons however only revealed drained class to have a significantly lower hydrophytic cover than restored age class 6-10 years ($p = 0.039$) (Figure 3.3b).

3.4.3 Native and Non-native Species

Mean percentage of native species was 64.36 (SD = ±18.21). The drained class had a lower mean percentage of native species (16.41, SD = ±12.26) in comparison to restored age classes 3-5 (68.05, SD = ±8.90), 6-10 (66.34, SD = ±6.38), 11-15 (65.56, SD = ±4.21), >20 (69.18, SD = ±6.97), Nat(Agr) (77.56, SD = ±5.22), and Nat(Pr) (75.46, SD = ±4.63) (Figure 3.4a). ANCOVA was not conducted on percentage of native species because of the significant interaction effect by area ($F_{(6,15)} = 2.930$, $p = 0.043$, partial $\eta^2 = 0.540$, observed power = 0.733).

Mean percent-cover of native species was 40.93 (SD = ±21.62). The percent-cover of native species showed trends similar to hydrophyte cover. ANCOVA results

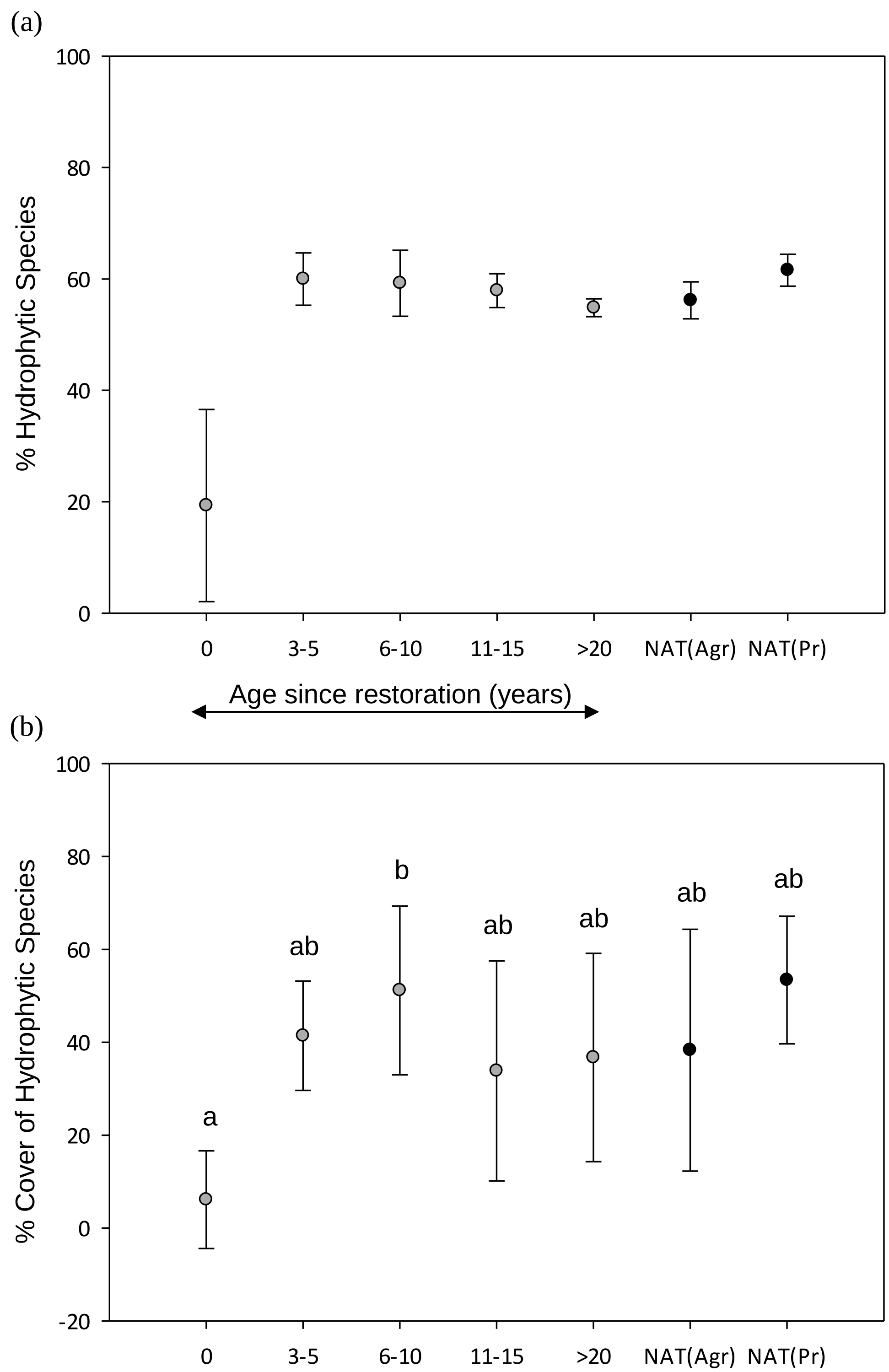

Figure 3.3 Mean ± SD (a) percentage of hydrophytic species, and (b) percent-cover of hydrophytic species across a chronosequence of restored wetlands. Age 0 represents drained wetlands. Natural wetlands are represented by black circles. Letters indicate significant differences.

confirmed a statistically significant difference in percent-cover of native species among age classes and natural wetlands ($F_{(6,21)} = 3.293$, $p = 0.019$, partial $\eta^2 = 0.485$, observed power = 0.838). Pairwise comparisons however only revealed drained class to have a significantly lower native cover than restored age class 6-10 years ($p = 0.010$) (Figure 3.4b).

Mean percentage of non-native species was 27.08 (SD = ±14.48). The drained class had a significantly higher mean percentage of non-native species (63.83, SD = ±12.95) in comparison to restored age classes 3-5 (23.86, SD = ±6.88), 6-10 (26.55, SD = ±4.15), 11-15 (26.43, SD = ±3.91), >20 (23.97, SD = ±7.21), Nat(Agr) (15.09, SD = ±5.39), and Nat(Pr) (18.52, SD = ±5.53). ANCOVA results confirmed a statistically significant difference in percentage of non-native species among age classes and natural wetlands ($F_{(6,21)} = 14.855$, $p < 0.0001$, partial $\eta^2 = 0.809$, observed power = 1). Pairwise comparisons revealed drained class to have a significantly higher percentage of non-native species than all other age classes, Nat(Agr), and Nat(Pr) (age classes 3-5 ($p < 0.0001$), 6-10 ($p < 0.0001$), 11-15 ($p < 0.0001$), >20 ($p < 0.0001$), and Nat(Agr) ($p < 0.0001$)) (Figure 3.5a).

Mean percent-cover of non-native species was 13.29 (SD = ±7.77). The mean percent-cover of non-native species was 16.89 (SD = ±1.88), 13.99 (SD = ±13.46), 11.56 (SD = ±7.90), 12.29 (SD = ±6.94), 13.65 (SD = ±7.40), 9.89 (SD = ±8.28), and 14.99 (SD = ±6.28) for age class 0, 3-5, 6-10, 11-15, >20, Nat(Agr), and Nat(Pr), respectively. ANCOVA results confirmed that no statistically significant difference existed among different age classes and natural wetlands ($F_{(6,21)} = 0.270$, $p = 0.945$, partial $\eta^2 = 0.072$, observed power = 0.105) (Figure 3.5b).

3.4.4 Sensitive Species

Five sensitive species were found in sampled wetlands. These included *Anemone virginiana* var. alba, *Juncus confusus*, and *Ranunculus uncinatus* found in restored sites (ROP1, BOW1, and OZM1) belonging to the youngest (3-5 years) restored age class, and

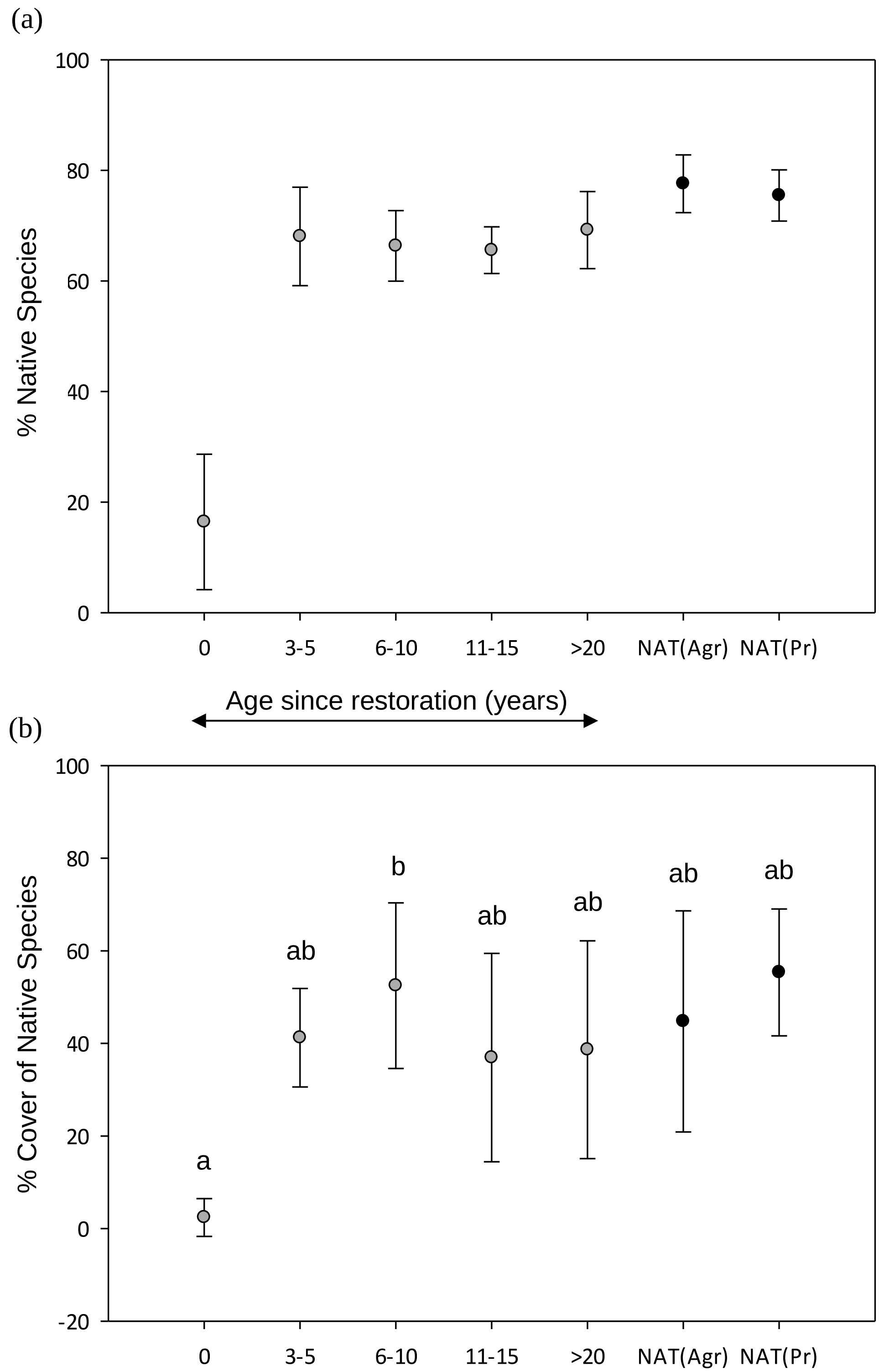

Figure 3.4 Mean ± SD (a) percentage of native species, and (b) percent-cover of native species across a chronosequence of restored wetlands. Age 0 represents drained wetlands. Natural wetlands are represented by black circles. Letters indicate significant differences.

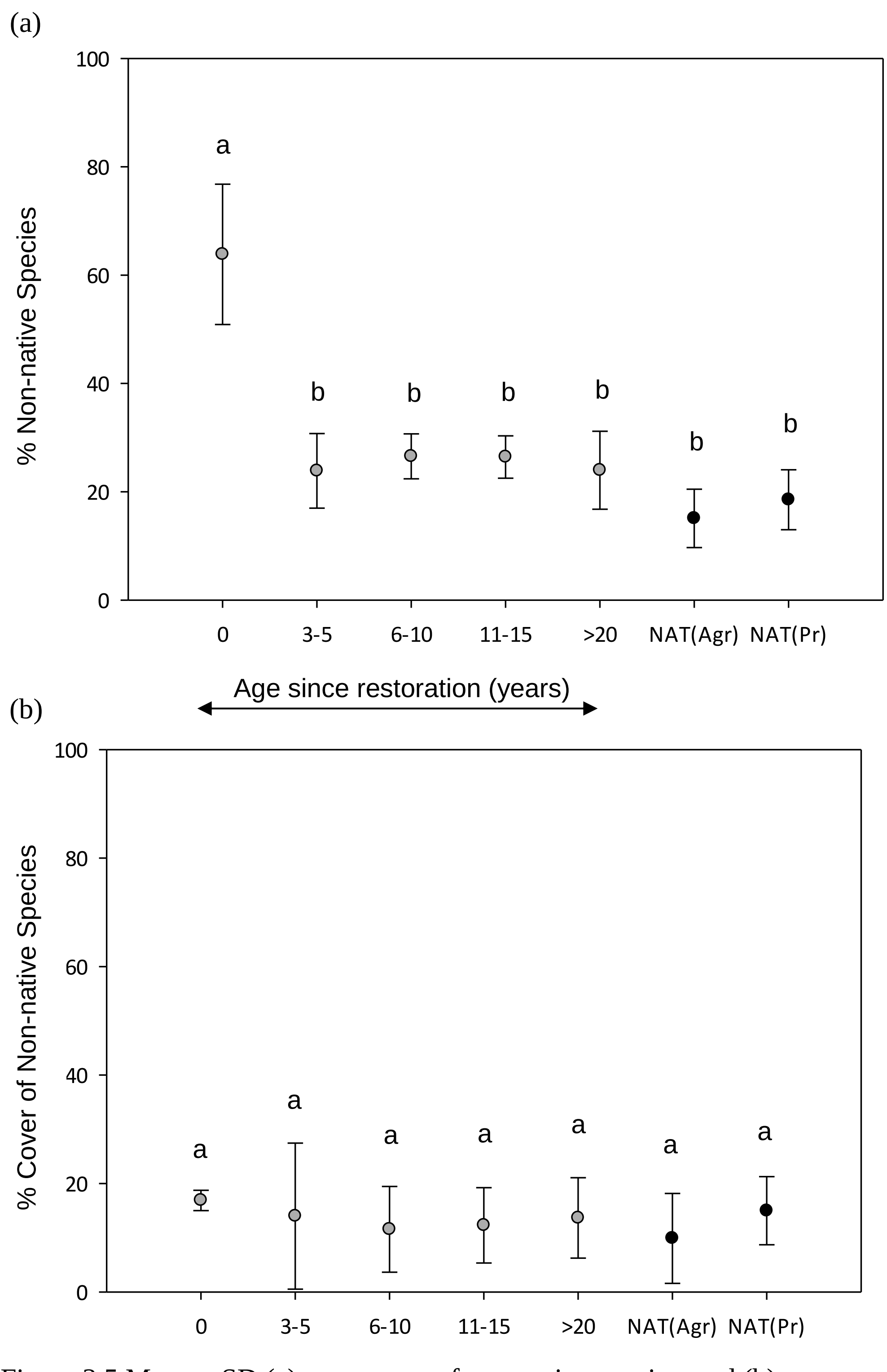

Figure 3.5 Mean ± SD (a) percentage of non-native species, and (b) percent-cover of non-native species across a chronosequence of restored wetlands. Age 0 represents drained wetlands. Natural wetlands are represented by black circles. Letters indicate significant differences.

Anemone virginiana var. alba, *Lonicera villosa*, *Lycopus asper*, and *Ranunculus uncinatus* found in natural wetlands (INT2, CLBID8, CLBIM1, and CLBRD3). All sensitive species are native-hydrophytes except *Anemone virginiana* var. alba which is an upland species.

3.4.5 Community Composition

Three main clusters were identified in the NMDS ordination of community composition as grouped by age classes, Nat(Agr), and Nat(Pr) (Figure 3.6). Each non-overlapping and widely separated cluster comprises sites with similar community composition. The drained class separated from others along the first NMDS axis, Nat(Pr) separated along the second NMDS axis, whereas restored age classes showed convergence in community composition to Nat(Agr) as indicated by their proximity to each other and overlapping clusters in the ordination. This dissimilarity in community composition was confirmed by PERMANOVA which showed a statistically significant difference (pseudo-$F_{(6,22)} = 3.63$, $p = 0.0001$) among different age classes and natural wetlands. Pairwise comparisons further indicated that the drained class was significantly dissimilar in community composition from others (age classes 3-5 ($p = 0.004$), 6-10 ($p = 0.003$), 11-15 ($p = 0.005$), >20 years ($p = 0.016$), Nat(Agr) ($p = 0.014$), and Nat(Pr) ($p < 0.001$)). Likewise, Nat(Pr) was also significantly dissimilar from age classes 3-5 ($p = 0.018$), 6-10 ($p = 0.004$), 11-15 ($p = 0.007$), >20 ($p = 0.013$), and Nat(Agr) ($p = 0.045$).

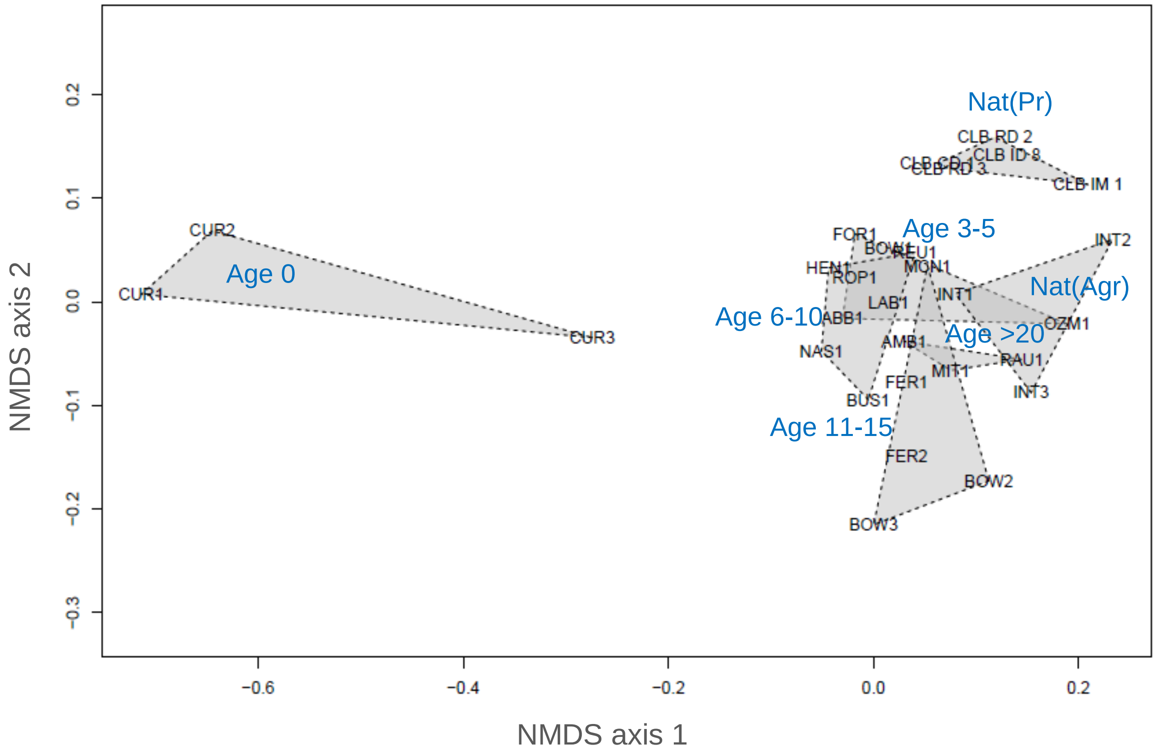

Figure 3.6 Non-metric multidimensional scaling (NMDS) of community composition as grouped by age classes, Nat(Agr), and Nat(Pr) (Stress = 0.15).

4 Discussion

The Prairie Pothole Region is comprised of wetlands of varying water permanence. Aronson & Galatowitsch (2008) recommended that recovery of prairie wetlands can be improved by focusing on all wetlands, irrespective of their water permanence, as together they add to the landscape-level integrity of the ecosystem. However, previous studies in the Canadian prairies have focused on recovery assessments of semi-permanent and permanent prairie wetlands (e.g., Bortolotti et al., 2016; Wilson et al., 2013; Wilson & Bayley, 2012; Forrest, 2010), and often ignored smaller wetlands which have low water permanence. This study fills this gap in knowledge by assessing vegetation recovery in temporary and seasonal prairie wetlands following restoration efforts and providing a more realistic assessment of the potential for wetland restoration within the Parkland ecoregion of the Canadian prairies.

4.1 Variability in Vegetation Associated with Wetland Morphometrics

Not surprisingly, wetland area, perimeter-to-area ratio, and slope were found to be associated with vegetation diversity in prairie wetlands (Table 3.1, Figure 3.1). A larger area typically provides more habitat heterogeneity and thus supports a wider variety of plant species (Mullhouse & Galatowitsch, 2003; Aronson & Galatowitsch, 2008). It also increases the likelihood that a wetland will receive more plant propagules and seeds from nearby sources, especially those with poor dispersal limits, as well as provides opportunities for within site dispersal, thereby adding to both species richness and cover (Cook et al., 2005; Mullhouse & Galatowitsch, 2003; Møller & Rordam, 1985). However, wetlands restored within the Canadian prairies are usually small, positioned in the agricultural landscape, and isolated from high-integrity natural wetlands. Thus, even in relatively larger wetlands, there are limits to the potential for recovery of vegetation in restored wetlands, as found in this study.

As expected, larger perimeter-to-area ratio was positively correlated to percentage of non-native species and negatively to native species (Figure 3.1). Since adjacent areas surrounding wetlands on the agricultural landscape often contains many non-native and opportunistic upland species (see Harker et al., 2009), it overwhelms the potential establishment of native-hydrophytes (Young et al., 2001). Species like *Bromus inermis* (a non-native upland species found in 24 sites), *Cirsium arvense* (a non-native upland species found in all 29 sites), and *Sonchus arvensis* (a non-native hydrophytic species found in 25 sites) are particularly detrimental as they aggressively spread *via* vegetative growth forming dense colonies, in addition to, their seeds being dispersed to larger distances (Otfinowski et al., 2007, Lemna & Messersmith, 1990; Moore, 1975). Thus, a higher perimeter-to-area ratio negatively affects native species in prairie wetlands as it provides more entry points for non-native species to invade the ecosystem, thereby decreasing overall species richness and affecting vegetation community composition.

Similarly, steeper slopes were found to be positively associated with high percentage and cover of non-native species in prairie wetlands. This is because steeper slopes generally undergo rapid changes in soil moisture during variable hydroperiods, a characteristic feature of prairie wetlands which cycles through periods of drought and deluge, that increases the wetland’s susceptibility to upland opportunistic and non-native species (Wilson et al., 2013; Wilson & Bayley, 2012; Forrest, 2010). Hence, efforts to restore wetlands should focus on gentler slopes as this will be advantageous to control spread of non-native species in restored wetlands and ensure successful recovery of vegetation.

4.2 Recovery of Vegetation Across a Restoration Chronosequence

Based on analyzed vegetation diversity metrics (observed species richness, estimated species richness, cover of hydrophytic and native species, and community composition) recovery of vegetation was achieved within 3-5 years of hydrological restoration in prairie wetlands when compared to low-integrity natural wetlands. This timeline is comparable to recovery of vegetation in restored wetlands of similar ages in other studies conducted in the Parkland ecoregion of the Canadian prairies (e.g., Bortolotti et al., 2016; Wilson et al., 2013; Wilson & Bayley, 2012; Puchniak, 2002).

Drained wetlands represented by age class 0 provided a baseline marking the initial conditions of a wetland undergoing recovery. As expected, drained wetlands differed significantly in vegetation community composition from restored and natural wetlands (Figure 3.6). However, they did not differ significantly from age class 11-15 years in terms of observed species richness due to some sites having low species richness within that class (Figure 3.2a). This decline in species richness in age class 11-15 years could reflect a part of successional trajectory in which wetlands post-restoration experience a gradual decline in species richness, after an initial influx of species, followed by stabilization as wetland species start to accumulate and gain dominance (see Noon, 1996). Likewise, drained class did not differ significantly in estimated species richness from restored (except age class 3-5) and low-integrity natural wetlands because this metric could only be compared at the maximum sampling effort in drained wetlands (Figure 3.2b). Because drained wetlands had minimal vegetation cover only 6 quadrats were sampled and species richness was extrapolated to double the quadrats sampled (6*2 = 12) to allow meaningful comparisons to be made at the sampling effort of 12 across all sites. It is speculated that if species richness could be estimated at a higher sampling effort, a significant difference would become evident as total number of observed species richness was low in drained wetlands. Also, the non-significant difference in percent-cover of hydrophytes of drained wetlands to restored (except age class 6-10) and low-integrity natural wetlands may be due to presence of non-native hydrophytic species *Sonchus arvensis* and *Plantago major* in one of the drained sites that had a small portion of wet area (Figure 3.3b). Likewise, the non-significant difference in percent-cover of natives of drained wetlands to restored (except age class 6-10) and low-integrity natural wetlands may be due to some sites having very low native species cover, which lowered the class mean to show any significant differences (Figure 3.4b).

Older restored sites (>20 years) were expected to have a higher wetland species diversity than younger sites (3-5 years). Since no significant differences were found in vegetation diversity metrics (excluding percentage of hydrophytic and native species which could not be evaluated as they varied with both age and area of the wetland) across the chronosequence of restored wetlands, older restored sites did not have a higher species diversity than younger sites (Figure 3.2, 3.3b, 3.4b, 3.5). Furthermore, there was no significant difference between restored wetlands, and low- and high-integrity natural wetlands, except for when restored wetlands continually maintained a significantly low species richness at quadrat scale after 3-5 yeas of restoration along with low-integrity natural wetlands (Figure 3.2, 3.3b, 3.4b, 3.5).

Following restoration, wetlands generally undergo a period of "self-design" (Mitsch & Wilson, 1996) and "self-organization" (Odum, 1989) during which succession takes place. Temporary and seasonal wetlands often experience rapid species accumulation and extinction rates due to their variable hydroperiod (Aronson & Galatowitsch, 2008). However, species colonize at different rates owing to dispersal limitations, on-site constraints, and landscape isolation (Galatowitsch, 2006; Mulhouse & Galatowitsch, 2003), which may be a reason why restored sites after 3-5 years of restoration along with low-integrity natural wetlands continued to fail to achieve species richness similarity at quadrat scale to high-integrity natural wetlands.

Restored wetlands closely resemble the vegetation community composition of nearby low-integrity natural wetlands (Figure 3.6). However, all wetlands on the agricultural landscape, irrespective of their type (i.e., drained, restored or natural), had a vegetation community that differed from high-integrity natural wetlands. Considering that high-integrity natural wetlands represented the least disturbed wetland conditions, this difference in vegetation community suggests that restoration efforts are failing to achieve maximum restoration potential. Of the 32 species that were completely absent in wetlands on the agricultural landscape, 23 were native-hydrophytic species. These included but were not limited to *Agrostis scabra*, *Cardamine pensylvanica*, *Carex diandra*, *Carex disperma*, *Castilleja miniata*, *Comarum palustre*, *Geum macrophyllum*, *Geum rivale*, *Lysimachia ciliata*, *Ribes glandulosum*, *Ribes hudsonianum*, *Sparganium eurycarpum*, *Sphenopholis intermedia*, *Stachys palustris*, *Thalictrum venulosum*, and

Viola renifolia. Though these species are not currently at risk, continued loss of high-integrity natural wetlands will likely result in a depletion of seedbanks and habitat availability in the future.

Significant differences in vegetation community composition as indicated by PERMANOVA may have emerged due to limitations of dispersal of individual species, competition of non-native species, and a depauperate native seedbank. Many native-hydrophytes (e.g., *Cardamine pensylvanica*, *Lysimachia ciliata*) that belong to sedge meadow, wet prairie, and woody perennial plant communities generally have low colonization efficiency due to lack of dispersal vectors (Aronson & Galatowitsch, 2008; Galatowitsch & van der Valk, 1996) which may have precluded their establishment in wetlands on the agricultural landscape. In addition, presence of non-native perennial species, such as *Bromus inermis*, *Cirsium arvense*, *Sonchus arvensis*, and *Taraxacum officinale*, in restored wetlands, especially early during the recovery period (Figure 3.5), may have precluded establishment of some native-hydrophytes found in high-integrity natural wetlands (Young et al., 2001; see Otfinowski et al., 2007; Stewart-Wade et al. 2002; Lemna & Messersmith, 1990; Moore, 1975 for invasion by these species). Furthermore, Weinhold & van der Valk (1989) noted that native seedbank density and diversity declines with time in drained wetlands, which affects their recovery potential. For example, seeds of *Carex sp.* survive up to 40 years whereas seeds of *Sparganium eurycarpum* only survive up to 20 years in a drained wetland. Thus, non-establishment of certain native-hydrophytes indicates an absence of a viable seedbank in wetlands on the agriculture landscape. In a fragmented and isolated landscape, the effect of dispersal limitation, competition of non-natives species, and a depauperate seedbank can become more severe (Galatowitsch, 2006; Mulhouse & Galatowitsch, 2003). However, the extent and impact of these probable barriers to successful recovery can only be confirmed by a detailed study on fragmentation of seedbanks within the landscape (i.e., possible isolation) and persistence of seedbanks in the Canadian prairies.

Sensitive species were observed in restored wetlands, but these became absent as the sites aged, possibly reflecting the absence of favorable habitat conditions for them to survive and grow. However, no definitive argument can be made as to what is favoring or preventing the establishment of sensitive species in sites because their presence was strictly recorded on an observational basis in this study and factors affecting their establishment were not studied.

What constitutes a restoration success ultimately comes down to how it is specified in restoration policies. The Alberta Wetland Policy considers restoration success to be “re-establishment of natural hydrology, vegetation, and wetland processes within a previously drained wetland” (Government of Alberta, 2016). However, this definition explicitly ignores the integrity of natural wetlands. As found in this study, restoration success differed when compared to low- and high-integrity natural wetlands. It was successful when compared to low-integrity natural wetlands however it failed to maintain species richness (at quadrat scale) and community composition of high-integrity natural wetlands. Restoration efforts that aim to resemble natural wetlands of low-integrity are problematic, as it represents a slippery slope of diminished or diminishing restoration targets (Kentula, 2000). Hence, care must be exercised when setting restoration goals and success criteria.

Interestingly, unlike the Canadian prairies, restorations have been quite unsuccessful in the US where restored prairie wetlands have failed to establish expansive sedge meadow and wet prairie plant communities (e.g., Aronson & Galatowitsch, 2008; Mullhouse & Galatowitsch, 2003; Galatowitsch & van der Valk, 1996). This geographic difference in recovery is credited to differences in drainage history and climate conditions (Bortolotti et al., 2016; Puchniak, 2002). The duration of drainage negatively impacts native seedbank and wetland hydrology (Weinhold & van der Valk, 1989). The long history of intensive agricultural drainage in the US since the 1900s (USDA, 1987) have significantly altered the landscape resulting in a decreased number of natural wetlands that can serve as viable seedbanks or propagule sources of native-hydrophytes. In contrast, agricultural drainage in Alberta only became an intensive activity beginning in the late 1950s (Jutras & Broughton, 2013). Puchniak (2002) noted that restored wetlands in Canada and the US differed in their drainage method, mostly ditch-drained in Canada and tile-drained in the US (e.g., Galatowitsch & van der Valk, 1996; Galatowitsch & van der Valk, 1995; Weinhold & van der Valk, 1989). This difference in drainage method may also be a contributing factor to the observed geographic difference in recovery, as tile-drained sites have lower vegetation recovery potential than ditch-drained sites (Galatowitsch & van der Valk, 1995). However, drainage method no longer influences recovery potential after a decade of restoration (Mullhouse & Galatowitsch, 2003). Finally, climatic conditions vary a lot across the Prairie Pothole Region with a strong north-south temperature and east-west precipitation gradient (Johnson et al., 2005), which regulates wetland hydroperiod and growth of many plant species. Though Puchniak (2002) accounted for the fact that climate favored recovery of vegetation in the Canadian prairies, Johnson & Poiani, (2016) concluded that the current warming of the Canadian prairies that resulted in decreased precipitation will substantially decrease its recovery potential in the future.

4.3 Implications for Wetland Restoration

This study indicates that within the Canadian prairies, restored wetlands have vegetation community composition similar to low-integrity natural wetlands on the agriculture lands but not high-integrity natural wetlands in protected areas. This implies that although abiotic barriers to restore hydrology in these previously ditch-drained wetlands were overcome, recovery will further require vegetation manipulation to ensure the return of many sensitive and native species in wetlands.

The differences in vegetation community composition of restored and low-integrity natural wetlands when compared to high-integrity natural wetlands may be attributed to depauperate seedbanks and presence of non-native species in restored wetlands, which precludes the establishment of native-hydrophytes. To overcome this, control of non-native species along with active plantation of missing native-hydrophytes is advised. Restoration strategies that target larger wetlands with lower perimeter-to-area ratios and gentler slopes will likely lead to greater potential of recovery of vegetation in restored wetlands. However, these strategies should be exercised carefully against a backdrop of increasing risk of landscape isolation, where restoring many smaller wetlands with intact edges nearby high-integrity natural wetlands may be better than restoring larger isolated wetlands with degraded edges (Kirkman et al., 2012; Ries et al., 2004). In this study, the influence of landscape isolation on the recovery potential of drained wetlands could not be considered due to lack of availability of a precise and accurate wetland inventory for the region.

Even though recovery of most vegetation-based metrics was achieved within 3-5 years of restoration, monitoring of restored wetlands should extend beyond a typical period of 5 years to track recovery path and ensure continued restoration success. This was evident when restored wetlands continually failed to achieve species richness similarity (at quadrat scale) after 3-5 years of restoration to that of high-integrity natural wetlands. Also, given that many native-hydrophytic species were absent from wetlands on the agriculture landscape, and that restoration efforts will likely suffer under future climate conditions, it is advised that more natural wetlands should be protected from both landscape fragmentation and land-conversion activities.

5 Conclusion

Wetlands are important components of our landscape providing many ecosystem services to society and habitat to a wide variety of flora and fauna. The wetlands of the Prairie Pothole Region are rich hotspots of biodiversity supporting nearly half of North America's waterfowl population (Environment Canada, 2013; Galatowitsch & van der Valk, 1998; Batt et al., 1989). Unfortunately, these wetlands have been most vulnerable to land-conversion activities (McLaughlin et al., 2014; Kennedy & Mayer, 2002). As such, many government policies mandating restoration to mitigate wetland loss and degradation have been adopted. However, the continued practice of restoring these ecosystems under such policies warrants the need to assess their success to mimic natural wetlands.

5.1 Research Findings

This study determined the potential rate and success of vegetation recovery in restored temporary and seasonal prairie wetlands located in the Central Parkland ecoregion of Alberta, Canada. The study found that various wetland morphometrics influence vegetation diversity, such that larger areas, lower perimeter-to-area ratio, and gentler slopes can favor greater recovery of native-hydrophytic vegetation in restored wetlands. The study also found that restored wetlands resemble vegetation diversity and community composition of low-integrity natural wetlands that occur on agricultural landscape within 3-5 years of restoration. However, they maintain a significantly low species richness at quadrat scale and differ in community composition when compared to high-integrity natural wetlands that occur on protected landscape. This failure of restored wetlands to resemble high-integrity natural wetlands highlights the loss of many native species from agricultural landscape and warrants the need for improved region-specific wetland restoration actions.

5.2 Research Significance

This study provides a scientific evidence to use wetland restoration as a successful compensation for wetland loss and degradation in the Parkland ecoregion of the Canadian prairies. While Wilson & Bayley (2012) have already highlighted the differential success of restoration when compared to low- and high-integrity natural wetlands, this study complements their results by investigating the effect of wetland morphometry on vegetation, the region-specific rate of vegetation recovery, and the differences in community composition among temporary and seasonal wetlands of different types.

The study contributes to an understanding of how restored wetlands perform upon establishment and as they age. In doing so, it discusses the implications for wetland restoration, and provides key recommendations to improve wetland restoration and management actions. First, it recommends measures that control non-native species such as *Cirsium arvense* and *Sonchus arvensis* (i.e., weeding) and promote native-hydrophytes that are completely absent from wetlands on the agricultural landscape (i.e., plantings). Second, it recommends restoration strategies to focus on larger wetlands with lower perimeter-to-area ratio and gentler slopes to increase the probability of restoration success. Finally, monitoring of restored wetlands beyond a typical 5-year period is recommended to ensure continued restoration success. In addition to these recommendations, the study also vouches for the continued protection of high-integrity natural wetlands to prevent the further loss of many sensitive and or native species.

5.3 Future Research Direction

A similar study across different ecoregions within the Canadian prairies would help to provide a more robust assessment of wetland recovery and restoration success. Future studies on presence of native seedbanks and landscape isolation will supplement the results in this study by identifying specific barriers to wetland restoration and forming a more detailed restoration response for the Parkland ecoregion.

Achieving vegetation similarity to natural wetlands is generally not enough to confidently conclude restoration success because achieving structural similarity may not be same as achieving functional similarity or *vice-versa*. It is thus necessary that restored wetlands must also be evaluated on their ability to provide ecosystem services. Many studies have found strong links between wetland structure (i.e., vegetation community composition) and wetland function (e.g., nutrient cycling) (Ehrenfeld, 2003; Hooper & Vitousek, 1997; Lauenroth et al., 1993). A comprehensive study evaluating these links will likely supplement this study and benefit future wetland restoration efforts within the Canadian prairies. The plant species data collected in this study may serve as a starting point for such a type of study.

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Appendices

Appendix A. List of metrics to assess vegetation in an ecosystem. References are provided where each metric has been used to assess vegetation.

Vegetation Measure	Method	References
Species Richness	Number of species $S=cA^z$ where S is species richness, A is area, c is constant and z is slope	(DeBerry & Perry, 2012) (Spieles, 2005) (Stefanik & Mitsch, 2012) (Moreno-Mateos et al., 2012) (Seabloom & van der Valk, 2003) (Ho & Richardson, 2013) (Miller & Wardrop, 2006) (Kellogg & Bridgham, 2002) (Spieles et al., 2006) (Wilson & Bayley, 2012) (Morgan & Short, 2002) (Highland et al., 2015) (Zhang et al., 2015) (McLachlan & Knispel, 2004) (De Steven et al., 2010) (Meyer et al., 2010) (Wentzell et al., 2016) (Lopez & Fennessy, 2002) (Bourdaghs et al., 2006) (Puchniak, 2002)
Carex Richness	Number of *Carex* species	(Mathews et al., 2009)
Typha Latifolia Richness	Stem count of *Typha latifolia*	(Wilson & Bayley, 2012)
Community (or Group e.g. life history or taxonomic groups) Richness	Number, Proportion, Percentage of different communities (or groups)	(Stefanik & Mitsch, 2012) (Mathews et al., 2009) (Wilson & Bayley, 2012) (De Steven et al., 2010)
Effective Species Richness	Reciprocal of Simpson's Diversity Index	(McLachlan & Knispel, 2004)
Species (or Group e.g. life history or taxonomic groups) Cover and Abundance	Percent cover Plant abundance Relative cover Relative abundance Percent cover of *Carex* species	(Moreno-Mateos et al., 2012) (Seabloom & van der Valk, 2003) (Bortolotti et al., 2016) (Wilson & Bayley, 2012) (Morgan & Short, 2002) (De Steven et al., 2010) (Meyer et al., 2010) (Aronson & Galatowitsch, 2008) (Puchniak, 2002) (Ho & Richardson, 2013)

Species Evenness	Pielou's evenness Reciprocal of Simpson's Diversity Index/ Total number of species	(Zhang et al., 2015) (McLachlan & Knispel, 2004) (Wentzell et al., 2016)
Native Species	Number of native species Number of native genera Proportion of native species Percentage of native species Percent cover of native species Percent cover of native perennials Effective native species richness	(Mathews et al., 2009) (Miller & Wardrop, 2006) (Wilson & Bayley, 2012) (Highland et al., 2015) (McLachlan & Knispel, 2004) (Lopez & Fennessy, 2002) (Bourdaghs et al., 2006)
Non-native Species	Number of non-native species Percentage of non-native species by cover Percentage of non-native species by frequency Effective non-native species richness	(Spieles, 2005) (Ho & Richardson, 2013) (Miller & Wardrop, 2006) (Spieles et al., 2006) (Wilson & Bayley, 2012) (Seabloom & van der Valk, 2003) (Highland et al., 2015) (McLachlan & Knispel, 2004)
Rare Species	Number of rare species	(Aronson & Galatowitsch, 2008)
Species Diversity	Shannon's Diversity Index Simpson's Diversity Index	(DeBerry & Perry, 2012) (Stefanik & Mitsch, 2012) (Kellogg & Bridgham, 2002) (Highland et al., 2015) (Zhang et al., 2015) (De Steven et al., 2010) (Meyer et al., 2010) (Wentzell et al., 2016) (Puchniak, 2002)
Community Diversity Index (CDI)	$CDI = -\sum_{i=1}^{N} C_i \, (\ln C_i)$ where N is number of wetland communities and C is relative area of each community	(Stefanik & Mitsch, 2012)
Species Composition	Sorensen Similarity Index, Bray Curtis Dissimilarity, Mantel tests, Ordination	(DeBerry & Perry, 2012) (Mathews & Spyreas, 2011) (Seabloom & van der Valk, 2003) (Ho & Richardson, 2013) (McLachlan & Knispel, 2004) (Meyer et al., 2010) (Aronson &

		Galatowitsch, 2008) (Wentzell et al., 2016) (Puchniak, 2002)
Importance Values (IV)	IV of each species, perennials, native species, hydrophytic species	(DeBerry & Perry, 2012) (Mathews et al., 2009)
Species Dominance	50:20 rule to mean IV	(DeBerry & Perry, 2012)
Prevalence Index	Weighted average of wetland indicator status and percent cover	(Spieles, 2005) (Spieles et al., 2006) (Meyer et al., 2010)
Mean coefficient of conservatism (C)	Mean C = $\sum c / N$ where c is the coefficient of conservatism score of each species and N is total species number	(Mathews et al., 2009) (Miller & Wardrop, 2006) (Bourdaghs et al., 2006)
Conservative Richness	Coefficient of Conservatism > 5	(Mathews et al., 2009)
Floristic Quality Index (FQI)	FQI = mean C $\sqrt{N}$ where C is mean coefficient of conservatism for all species and N is total species number	(Mathews et al., 2009) (Wentzell et al., 2016) (Bourdaghs et al., 2006)
Floristic Quality Adjustment Index (I)	I $=\sum(CC_i\sqrt{N})$ where CC_i is coefficient of conservatism for all species and N is the number of native species	(Stefanik & Mitsch, 2012) (Miller & Wardrop, 2006) (Spieles et al., 2006) (Wilson & Bayley, 2012) (Wentzell et al., 2016) (Lopez & Fennessy, 2002)
Adjusted FQAI (I′)	I′ = (mean C/ 10 * $\sqrt{N}/\sqrt{(N+A)}$) *100 where C is mean coefficient of conservatism value of native species, N is number of native species and A is number of non-native species	(Miller & Wardrop, 2006) (Wilson et al., 2013)
Functional group richness/diversity/composition	Number of species in guild Percent cover of species in guild Ruderals: Interstitial: Matrix species	(Aronson & Galatowitsch, 2008) (Stefanik & Mitsch, 2012) (Kellogg & Bridgham, 2002) (Wilson & Bayley, 2012) (Zhang et al., 2015) (De Steven et al., 2010)

	Functional group richness Functional regularity Functional divergence	
Aboveground net primary productivity	WANPP = $\sum(A_i B_i)/E$ where A is area of specific community, B is average biomass of specific community and E is total area of emergent plant communities	(Stefanik & Mitsch, 2012)
Biomass		(Moreno-Mateos et al., 2012) (Kellogg & Bridgham, 2002) (Morgan & Short, 2002) (Lopez & Fennessy, 2002)

References for Appendix A

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Appendix B. Summary of wetland delineation based on vegetation, soil, and hydrology.

Site ID	**Delineation based on wetland vegetation, soil, and hydrology**	**Δ Area (ha)**
CUR1	Since the wetland was drained and cultivated throughout with *Brassica sp*., boundary delineation based on vegetation could not be achieved. Soil pits however confirmed presence of dark loamy soil with minor evidence of oxidized rhizospheres at approx. 30 cm. It was speculated that <1% of wetland would contain surface water only seasonally during the wettest time of the year. Due to the inability to confirm boundaries in the field, contour lines were generally followed.	-
CUR2	Since the wetland was drained and cultivated throughout with *Brassica sp*., boundary delineation based on vegetation could not be achieved. Soil pits taken at middle of the wetland confirmed absence of hydric soils. Therefore, consideration of site as drained wetland was solely based on personal communication with the landowner. It was speculated that 1-25% of wetland would contain surface water only seasonally during the wettest time of the year. Due to the inability to confirm boundaries in the field, contour lines were generally followed.	-
CUR3	Since the wetland was drained and cultivated throughout with *Brassica sp*., boundary delineation based on vegetation could not be achieved. Soil pits however confirmed presence of dark loamy soil with minor evidence of oxidized rhizospheres at approx. 30 cm. It was speculated that 25-50% of wetland would contain surface water only seasonally during the wettest time of the year. Due to the inability to confirm boundaries in the field, contour lines were generally followed.	-
FOR1	Wetland on an average had a slope of <1°. Delineation was easy due to presence of distinct vegetative boundary formed by *Medicago sativa* and *Bromus inermis* at the wetland-upland interface. Soil pits taken at the boundary were a mix of moist loamy and clayey soils with minor evidence of mottles. Approx. 5 cm of standing water was present in a small pool at the time of assessment however soil within the emergent zone was saturated. It was speculated that nearly 50-95% of wetland would contain surface water only seasonally during the wettest time of the year.	-0.63
ROP1	Wetland slope varied between 1-3°. Delineation was easy due to presence of distinct vegetative boundaries. *Bromus ciliatus* and *Poa palustris* dominated the wet meadow zone whereas *Bromus inermis*, *Medicago sativa*, *Taraxacum officinale*, and *Trifolium hybridum* dominated the wetland-upland interface. Soil pits taken within the wetland boundary confirmed presence of mottles in the black clayey soils which was otherwise absent in the soil	-0.41

	outside the boundary. Approx. 40 cm of standing water was present in the wetland at the time of assessment. It was speculated that nearly 50-95% of wetland would contain surface water only seasonally during the wettest time of the year.	
ABB1	Wetland slope varied between 1-5°. Delineation was easy due to presence of distinct vegetative boundaries. *Typha latifolia* encircled the emergent zone whereas *Cirsium arvense*, *Taraxacum officinale*, and *Trifolium hybridum* dominated the wetland-upland interface. In addition, small *Salix sp*. sparsely encircled the boundary which further aided in the delineation process. Soil pits taken within the wetland boundary confirmed presence of mottles in the dark loamy soils which was otherwise absent in the clayey soils outside the boundary. Approx. 20 cm of standing water was present in the emergent zone at the time of assessment. It was speculated that nearly 25-50% of wetland would contain surface water only seasonally during the wettest time of the year.	-0.59
BOW1	Wetland on an average had a slope of 2.5°. Delineation was easy due to presence of distinct vegetative boundaries. *Typha latifolia* encircled the emergent zone, *Poa plaustris* dominated the wet meadow zone whereas *Bromus inermis* and *Medicago sativa* dominated the wetland-upland interface. Soil pits taken within the wetland boundary confirmed presence of mottles in the dark loamy soils which was otherwise absent in the dry loamy soils outside the boundary. Approx. 30 cm of standing water was present in a small portion of the emergent zone at the time of assessment. It was speculated that nearly 50-95% of wetland would contain surface water only seasonally during the wettest time of the year.	-1.17
OZM1	Wetland slope varied between 1-2°. Delineation was easy due to presence of distinct vegetative boundaries. *Carex atherodes* and *Poa palustris* dominated the wetland whereas *Salix sp*., *Populus sp*., and *Rosa acicularis* formed an extensive riparian zone marking the wetland-upland interface. Soil pits taken within the wetland boundary confirmed presence of mottles, gleying, and oxidized rhizospheres in the loamy-clayey soils which was otherwise absent in the soil outside the boundary. Surface water was absent at the time of assessment. However, it was speculated that nearly 50-95% of wetland would contain surface water only seasonally during the wettest time of the year.	-0.20

LAB1	Wetland on an average had a slope of 2°. Vegetative boundaries were a bit fuzzy as upland weedy species such as *Cirsium arvense*, *Bromus inermis*, and *Taraxacum officinale* dominated the wet meadow zone. Soil pits taken at the fuzzy boundary confirmed presence of mottles in the clayey soils which were otherwise absent in the soil outside the boundary. Approx. 3 cm of standing water was present in the emergent zone at the time of assessment. It was speculated that >95% of wetland would contain surface water only seasonally during the wettest time of the year.	-0.33
BUS1	Wetland was in a prominent depression and had an average slope of 5°. However, the north-east side of the wetland was much steeper with an approx. 10° slope. Delineation was easy due to presence of distinct vegetative boundaries. *Typha latifolia* interspersed with *Salix sp.* encircled the emergent zone, *Carex atherodes* dominated the wet meadow zone whereas *Bromus inermis*, *Medicago sativa*, and *Cirsium arvense* dominated the wetland-upland interface. Soil was, in general, dark, dry, and crumbly but it contained mottles and oxidized rhizospheres within the wetland boundary. Surface water was absent at the time of assessment however soil within the emergent zone was saturated. It was speculated that nearly 50-95% of wetland would contain surface water only seasonally during the wettest time of the year.	-0.15
NASI	Wetland slope varied between 2-3°. Delineation was easy due to presence of distinct vegetative boundaries. *Agropyron sp*. dominated the wet meadow zone, *Salix sp.*, *Populus tremuloides*, and *Medicago sativa* dominated the wetland-upland interface in the north whereas *Bromus inermis* dominated the upland. Soil pits taken within the wetland boundary confirmed presence of mottles in the saturated soils which was otherwise absent in the clayey soils outside the boundary. Approx. 20 cm of standing water was present in a small pool at the time of assessment. It was speculated that nearly 50-95% of wetland would contain surface water only seasonally during the wettest time of the year.	-0.33
HEN1	Wetland on an average had a slope of <1°. Delineation was easy due to presence of distinct vegetative boundaries. *Carex atherodes*, *Carex bebbii*, and *Poa palustris* dominated the wetland whereas *Bromus inermis* dominated the upland. Soil pits taken within the wetland boundary confirmed presence of mottles and oxidized rhizospheres which were otherwise absent in the clayey soils outside the boundary.	-1.03

	Approx. 35 cm of standing water was present in the wetland at the time of assessment. It was speculated that nearly 50-95% of wetland would contain surface water only seasonally during the wettest time of the year.	
REU1	Wetland slope varied between 1-2°. Delineation was easy due to presence of distinct vegetative boundaries. *Carex atherodes* and *Poa palustris* dominated the wetland whereas *Bromus inermis* and *Poa pratensis* dominated the upland. Soil pits taken within the wetland boundary confirmed presence of mottles in the loamy soils which was otherwise absent in the soil outside the boundary. Surface water was absent at the time of assessment. However, it was speculated that nearly 25-50% of wetland would contain surface water only seasonally during the wettest time of the year.	-0.30
BOW2	Wetland on an average had a slope of <1°. Delineation was very difficult due to absence of distinct vegetative boundaries. *Agropyron sp*. dominated the entire wetland. Soil pits taken near the boundary contained dry sandy soils with no evidence of redoximorphic features. Surface water was absent at the time of assessment. However, it was speculated that nearly 50-95% of wetland would contain surface water only seasonally during the wettest time of the year. Hence, due to inadequacy of vegetation and soil pits to confirm wetland boundaries desktop delineation was closely followed while adjusting for small vegetation changes.	-0.23
BOW3	Wetland on an average had a slope of <1°. Delineation was easy due to presence of distinct vegetative boundaries. *Agropyron sp*. and *Poa pratensis* dominated the wetland whereas *Cirsium arvense*, *Sonchus arvensis*, and *Thlaspi arvense* dominated the wetland-upland interface. In addition, *Salix sp*. sparsely encircled the wetland which further aided in the delineation process. Soil pits taken within the wetland boundary confirmed presence of mottles which was otherwise absent in the moist sandy soils outside the boundary. Surface water was absent at the time of assessment however soil within the wetland was saturated. It was speculated that nearly 50-95% of wetland would contain surface water only seasonally during the wettest time of the year.	-0.18
FER1	Wetland slope varied between 1-3°. Vegetative boundaries were a bit fuzzy. *Carex atherodes*, *Carex utriculata*, and *Eleocharis palustris*	-0.01

	dominated the emergent zone, *Agropyron sp.* dominated the wet meadow zone, and *Bromus ciliatus* dominated the upland. Soil pits taken outside the fuzzy boundary were a mix of clayey and sandy soils with no evidence of redoximorphic features. Approx. 20 cm of standing water was present in the wetland at the time of assessment. It was speculated that nearly 50-95% of wetland would contain surface water only seasonally during the wettest time of the year.	
FER2	Wetland slope varied between 2-3°. Vegetative boundaries were a bit fuzzy. *Carex atherodes* dominated the emergent zone, *Agropyron sp.* dominated the wet meadow zone, and *Bromus ciliatus* dominated the upland. Soil pits taken within the wetland boundary confirmed presence of mottles and a thick organic layer which were otherwise absent in the clayey soils outside the boundary. Approx. 30 cm of standing water was present in the wetland at the time of assessment. It was speculated that nearly 50-95% of wetland would contain surface water only seasonally during the wettest time of the year.	-0.43
MCN1	Wetland on an average had a slope of <1°. Vegetative boundaries were a bit fuzzy due to the invasion by upland weedy species such as *Cirsium arvense* and *Bromus inermis* in the wet meadow zone. However, *Salix petiolaris* encircling the wetland aided in the delineation process. Soil pits taken at the fuzzy boundary were a mix of loamy and clayey soils with evidence of mottles which was otherwise absent in the soil outside the boundary. Approx. 4 cm of standing water was present in a small pool at the time of assessment. It was speculated that nearly 50-95% of wetland would contain surface water only seasonally during the wettest time of the year.	-0.06
RAU1	Wetland slope varied between 2-3°. Delineation was difficult due to the invasion by upland species such as *Bromus inermis*, *Poa pratensis*, and *Cirsium arvense* in the wet meadow zone. However, small patches of *Salix petiolaris* present in the east of wetland aided in the delineation process. Soil pits taken at the boundary were a mix of moist loamy and clayey soils with evidence of mottles which was otherwise absent in the soil outside the boundary. Approx. 30 cm of standing water was present in a small pool in the emergent zone at the time of assessment. It was speculated that nearly 50-95% of wetland would contain surface water only seasonally during the wettest time of the year.	-1.24

MIT1	Wetland on an average had a slope of 1°. Delineation was very difficult due to absence of distinct vegetative boundaries. *Agropyron sp*. and *Cirsium arvense* dominated the wet meadow zone. Soil pits taken within the wetland boundary confirmed presence of mottles in the saturated clayey soils which was otherwise absent in the soil outside the boundary. Approx. 15 cm of standing water was present in the wetland at the time of assessment. It was speculated that nearly 50-95% of wetland would contain surface water only seasonally during the wettest time of the year.	-0.04
AMB1	Approx. 5-10% of wetland had a prominent depression dominated by *Beckmannia syzigachne* and *Eleocharis palustris* but generally slope varied between 1-2°. Vegetative boundaries were a bit fuzzy as weeds invaded the wetland. *Cirsium arvense* dominated the upland followed by *Sonchus arvensis* and *Thalspi arvense*. On the other hand, *Carex atherodes* densely covered the entire wetland. Soil pits taken outside the fuzzy boundary consisted of dark and crumbly soils with no evidence of redoximorphic features. Surface water was absent at the time of assessment. However, it was speculated that nearly 50-95% of wetland would contain surface water only seasonally during the wettest time of the year.	-0.44
CLBCD1	Wetland on an average had a slope of 4° but was less steep in the north (2°). Vegetative boundaries were a bit fuzzy. *Carex atherodes* and *Poa palustris* dominated the wet meadow zone whereas *Bromus inermis* and *Phleum pratense* dominated the wetland-upland interface. Soil pits taken at the fuzzy boundary were a mix of loamy and sandy soils with evidence of mottles which was otherwise absent in the soil outside the boundary. Approx. 10 cm of standing water was present in the emergent zone at the time of assessment. It was speculated that nearly 25-50% of wetland would contain surface water only seasonally during the wettest time of the year.	-0.18
CLBID8	Wetland on an average had a slope of 1°. Delineation was very difficult due to extensive grazing within the wetland and in surrounding areas. No soil pits were taken and in general contour lines were followed. Surface water was absent at the time of assessment. However, it was speculated that nearly 1-25% of wetland would contain surface water only seasonally during the wettest time of the year. Hence, desktop delineation was closely followed while adjusting for	0.39

	small vegetation changes by excluding upland species such as *Bromus inermis* and *Rosa acicularis* from the wetland boundary.	
CLBIM1	Wetland slope varied between 1-3°. Delineation was easy due to presence of distinct vegetative boundaries. *Carex atherodes* dominated the wet meadow zone whereas *Salix sp*., *Populus sp*., *Rosa acicularis*, and *Rubus idaeus* formed an extensive riparian zone (except in the west) marking the wetland-upland interface. Soil pits taken within the wetland boundary confirmed presence of mottles in the loamy-clayey soils which was otherwise absent in the soil outside the boundary. Approx. 25 cm of standing water was present in the emergent zone at the time of assessment. It was speculated that nearly 50-95% of wetland would contain surface water only seasonally during the wettest time of the year.	0.19
CLBRD2	Wetland on an average had a slope of 1°. Delineation was easy due to presence of distinct vegetative boundaries. *Carex atherodes* and *Poa palustris* dominated the wetland whereas *Phleum pratense* and *Trifolium hybridum* dominated the wetland-upland interface. Soil pits taken within the wetland boundary confirmed presence of mottles in the loamy-clayey soils which was otherwise absent in the soil outside the boundary. Surface water was absent at the time of assessment. However, it was speculated that nearly 1-25% of wetland would contain surface water only seasonally during the wettest time of the year.	0.04
CLBRD3	Wetland slope varied between 2-3°. Delineation was easy due to presence of distinct vegetative boundaries. *Carex utriculata* dominated the wet meadow zone whereas *Bromus inermis*, *Dactylis glomerata*, *Phleum pratense*, and *Trifolium hybridum* dominated the wetland-upland interface. Soil pits taken within the wetland boundary confirmed presence of mottles in the clayey soils which was otherwise absent in the dry sandy soil outside the boundary Surface water was absent at the time of assessment. However, it was speculated that <1% of wetland would contain surface water only seasonally during the wettest time of the year.	-0.17
INT1	Wetland slope varied between 1-2°. Delineation was easy due to presence of distinct vegetative boundaries. *Carex atherodes* and *Phalaris arundinacea* dominated the wet meadow zone whereas *Bromus inermis* and *Poa pratensis* dominated the upland. Soil pits taken within the wetland boundary confirmed presence of	-0.98

	mottles and a thick organic layer which were otherwise absent in the clayey soils outside the boundary. Surface water was absent at the time of assessment however soil within the emergent zone was saturated. It was speculated that nearly 50-95% of wetland would contain surface water only seasonally during the wettest time of the year.	
INT2	Wetland slope generally varied between 1-2° but was much steeper in the west (5°). Delineation was difficult due to dominance of *Cirsium arvense* in the wet meadow zone, and presence of an extensive riparian zone surrounding the wetland. Soil pits taken near the fuzzy boundary confirmed presence of a thick organic layer and mottles which were otherwise absent in the soil outside the boundary. Surface water was absent at the time of assessment. However, it was speculated that nearly 50-95% of wetland would contain surface water only seasonally during the wettest time of the year.	-0.75
INT3	Wetland on an average had a slope of 1°. Delineation was easy due to presence of distinct vegetative boundaries. *Carex atherodes* dominated the wetland whereas *Poa pratensis* and *Bromus inermis* dominated the upland. In addition, many *Salix sp.* and *Populus tremuloides* encircled the boundary which further aided the delineation process. Soil pits taken at the boundary confirmed presence of mottles in the dark clayey soils which were otherwise absent in the soil outside the boundary. Approx. 45 cm of standing water was present in the wetland at the time of assessment. It was speculated that nearly 50-95% of wetland would contain surface water only seasonally during the wettest time of the year.	-1.51

Appendix C. List of 188 plant species found in 29 sampled study sites along with their nativity and wetland indicator status. Nomenclature closely follows Integrated Taxonomic Information System.

Species	**Common name**	**WIS**	**Origin**	**Family**
Achillea alpina	Siberian yarrow	NA	Native	Asteraceae
Achillea millefolium	Common yarrow	Upland	Native	Asteraceae
Actaea rubra	Red baneberry	Upland	Native	Ranunculaceae
Agrimonia striata	Woodland groovebur	Upland	Native	Rosaceae
Agropyron sp	NA	NA	NA	Poaceae
Agrostis scabra	Rough bentgrass	Hydrophyte	Native	Poaceae
Alisma plantago-aquatica	American water plantain	Hydrophyte	Native	Alismataceae
Alopecurus aequalis	Short-awn meadow-foxtail	Hydrophyte	Native	Poaceae
Alopecurus pratensis	Meadow-foxtail	Hydrophyte	non-Native	Poaceae
Anemone canadensis	Canadian anemone	Hydrophyte	Native	Ranunculaceae
Anemone virginiana var. alba	Tall thimbleweed	Upland	Native	Ranunculaceae
Antennaria sp	NA	NA	Native	Asteraceae
Aralia nudicaulis	Wild sarsaparilla	Upland	Native	Araliaceae
Arctium minus	Lesser burrdock	Upland	non-Native	Asteraceae
Artemisia absinthium	Common sagewort	NA	non-Native	Asteraceae
Artemisia sp	NA	NA	NA	Asteraceae
Beckmannia syzigachne	American slough grass	Hydrophyte	Native	Poaceae
Bidens cernua	Nodding burr-marigold	Hydrophyte	Native	Asteraceae
Bolboschoenus maritimus ssp. paludosus	Cosmopolitan bulrush	Hydrophyte	Native	Cyperaceae
Brassica napus	Turnip	NA	non-Native	Brassicaceae
Brassica sp	NA	NA	non-Native	Brassicaceae
Bromus ciliatus	Fringed brome	Hydrophyte	Native	Poaceae

Bromus inermis	Smooth brome	Upland	non-Native	Poaceae
Calamagrostis canadensis	Bluejoint reedgrass	Hydrophyte	Native	Poaceae
Calamagrostis stricta	Slimstem reedgrass	Hydrophyte	Native	Poaceae
Capsella bursa-pastoris	Shepherd's-purse	Upland	non-Native	Brassicaceae
Cardamine pensylvanica	Quaker bittercress	Hydrophyte	Native	Brassicaceae
Carex aquatilis	Leafy tussock sedge	Hydrophyte	Native	Cyperaceae
Carex atherodes	Wheat sedge	Hydrophyte	Native	Cyperaceae
Carex bebbii	Bebb's sedge	Hydrophyte	Native	Cyperaceae
Carex diandra	Lesser tussock sedge	Hydrophyte	Native	Cyperaceae
Carex disperma	Soft-leaf sedge	Hydrophyte	Native	Cyperaceae
Carex pellita	Wolly sedge	Hydrophyte	Native	Cyperaceae
Carex sp	NA	NA	Native	Cyperaceae
Carex sychnocephala	Many-head sedge	Hydrophyte	Native	Cyperaceae
Carex utriculata	Northwest territory sedge	Hydrophyte	Native	Cyperaceae
Castilleja miniata	Great red indian-paintbrush	Hydrophyte	Native	Scrophulariaceae
Cerastium nutans	Nodding mouse-ear chickweed	Hydrophyte	Native	Caryophyllaceae
Ceratophyllum demersum	Coon's-tail	Hydrophyte	Native	Ceratophyllaceae
Chamerion angustifolium	Fireweed	Hydrophyte	Native	Onagraceae
Chamerion latifolium	Dwarf fireweed	NA	Native	Onagraceae
Chenopodium album	Lamb's-quarters	Upland	non-Native	Chenopodiaceae
Cicuta maculata	Spotted water-hemlock	Hydrophyte	Native	Apiaceae
Cirsium arvense	Canadian thistle	Upland	non-Native	Asteraceae
Comarum palustre	Purple marshlocks	Hydrophyte	Native	Rosaceae
Cornus canadensis	Canadian bunchberry	Upland	Native	Cornaceae
Cornus sericea	Red osier-dogwood	Hydrophyte	Native	Cornaceae
Corylus cornuta	Beaked hazelnut	Upland	Native	Betulaceae
Crepis tectorum	Narrow-leaf hawk's-beard	NA	non-Native	Asteraceae

Dactylis glomerata	Orchard grass	Upland	non-Native	Poaceae
Deschampsia cespitosa	Tufted hairgrass	Hydrophyte	Native	Poaceae
Descurainia sophia	Flaxweed tansymustard	NA	non-Native	Brassicaceae
Eleocharis palustris	Common spike-rush	Hydrophyte	Native	Cyperaceae
Equisetum palustre	Marsh horsetail	Hydrophyte	Native	Equisetaceae
Equisetum pratense	Meadow horsetail	Hydrophyte	Native	Equisetaceae
Equisetum sylvaticum	Woodland horsetail	Hydrophyte	Native	Equisetaceae
Erigeron acris	Bitter fleabane	Hydrophyte	Native	Asteraceae
Erigeron philadelphicus	Philadelphia fleabane	Hydrophyte	Native	Asteraceae
Erigeron sp	NA	NA	Native	Asteraceae
Erucastrum gallicum	Common dog-mustard	NA	non-Native	Brassicaceae
Erysimum cheiranthoides	Worm-seed wallflower	Upland	Native	Brassicaceae
Fallopia convolvulus	Black-bindweed	Upland	non-Native	Polygonaceae
Festuca pratensis	Meadow fescue	Upland	non-Native	Poaceae
Fragaria virginiana	Virginia strawberry	Upland	Native	Rosaceae
Galeopsis tetrahit	Brittle-stem hemp-nettle	Upland	non-Native	Lamiaceae
Galium aparine	Sticky-willy	Upland	non-Native	Rubiaceae
Galium boreale	Northern bedstraw	Upland	Native	Rubiaceae
Galium labradoricum	Northern bog bedstraw	Hydrophyte	Native	Rubiaceae
Galium triflorum	Fragrant bedstraw	Upland	Native	Rubiaceae
Geranium sp	NA	NA	Native	Geraniaceae
Geum aleppicum	Yellow avens	Upland	Native	Rosaceae
Geum macrophyllum	Large-leaf avens	Hydrophyte	Native	Rosaceae
Geum rivale	Purple avens	Hydrophyte	Native	Rosaceae
Glyceria grandis	American manna grass	Hydrophyte	Native	Poaceae
Heracleum maximum	American cow parsnip	Hydrophyte	Native	Apiaceae
Hieracium umbellatum	Canadian hawkweed	NA	Native	Asteraceae

Hippuris vulgaris	Common mare's-tail	Hydrophyte	Native	Hippuridaceae
Hordeum jubatum	Foxtail barley	Hydrophyte	Native	Poaceae
Juncus balticus	Baltic rush	Hydrophyte	Native	Juncaceae
Juncus bufonius	Toad rush	Hydrophyte	Native	Juncaceae
Juncus confusus	Colorado rush	Hydrophyte	Native	Juncaceae
Lathyrus ochroleucus	Cream pea	NA	Native	Fabaceae
Lemna minor	Common duckweed	Hydrophyte	Native	Lemnaceae
Lemna trisulca	Ivy-leaf duckweed	Hydrophyte	Native	Lemnaceae
Linaria vulgaris	Common toadflax	NA	non-Native	Scrophulariaceae
Lonicera villosa	Mountain fly-honeysuckle	Hydrophyte	Native	Caprifoliaceae
Lycopus asper	Rough water-horehound	Hydrophyte	Native	Lamiaceae
Lysimachia ciliata	Fringed yellow-loosestrife	Hydrophyte	Native	Primulaceae
Lysimachia thyrsiflora	Tufted yellow-loosestrife	Hydrophyte	Native	Primulaceae
Maianthemum stellatum	Star-flowered Solomon's-seal	Upland	Native	Liliaceae
Medicago sativa	Alfalfa	Upland	non-Native	Fabaceae
Melilotus officinalis	Yellow sweet-clover	Upland	non-Native	Fabaceae
Mentha arvensis	Wild mint	Hydrophyte	Native	Lamiaceae
Mertensia paniculata	Tall bluebells	Hydrophyte	Native	Boraginaceae
Osmorhiza depauperata	Bluntseed sweetroot	NA	Native	Apiaceae
Penstemon procerus	Pincushion beardtongue	Upland	Native	Scrophulariaceae
Persicaria lapathifolia	Curlytop knotweed	Hydrophyte	Native	Polygonaceae
Petasites frigidus var. sagittatus	Arrowleaf sweet coltsfoot	Hydrophyte	Native	Asteraceae
Phalaris arundinacea	Reed canary grass	Hydrophyte	Native	Poaceae
Phleum pratense	Common timothy	Upland	non-Native	Poaceae
Plantago major	Great plantain	Hydrophyte	non-Native	Plantaginaceae
Poa palustris	Fowl blue grass	Hydrophyte	Native	Poaceae
Poa pratensis	Kentucky blue grass	Upland	Native	Poaceae

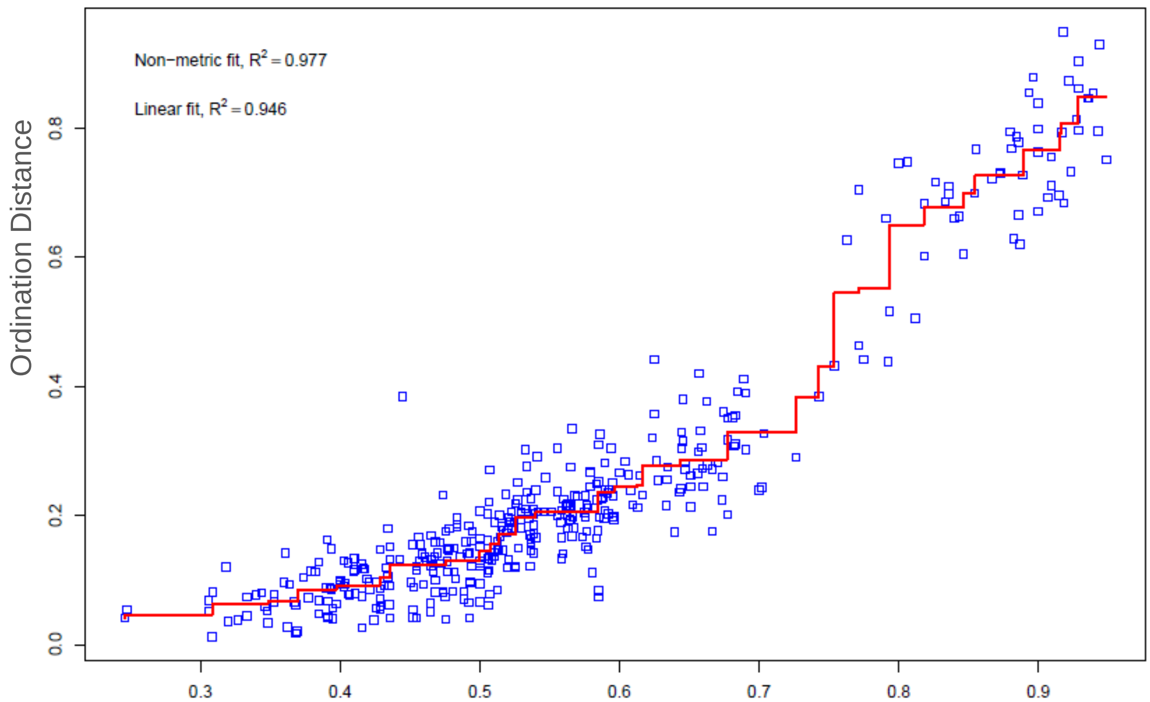
Polygonum aviculare	Yard knotweed	Upland	non-Native	Polygonaceae
Polygonum sp	NA	NA	NA	Polygonaceae
Populus balsamifera	Balsam poplar	Hydrophyte	Native	Salicaceae
Populus tremuloides	Quaking aspen	Hydrophyte	Native	Salicaceae
Potamogeton sp	NA	Hydrophyte	Native	Potamogetonaceae
Potentilla anserina	Silverweed cinquefoil	Hydrophyte	Native	Rosaceae
Potentilla gracilis	Graceful cinquefoil	Hydrophyte	Native	Rosaceae
Potentilla norvegica	Norwegian cinquefoil	Hydrophyte	Native	Rosaceae
Prunus virginiana	Choke cherry	Upland	Native	Rosaceae
Ranunculus gmelinii	Lesser yellow water buttercup	Hydrophyte	Native	Ranunculaceae
Ranunculus macounii	Macoun's buttercup	Hydrophyte	Native	Ranunculaceae
Ranunculus sceleratus	Cursed buttercup	Hydrophyte	Native	Ranunculaceae
Ranunculus uncinatus	Woodland buttercup	Hydrophyte	Native	Ranunculaceae
Ribes glandulosum	Skunk currant	Hydrophyte	Native	Grossulariaceae
Ribes hudsonianum	Northern black currant	Hydrophyte	Native	Grossulariaceae
Ribes oxyacanthoides	Canadian gooseberry	Upland	Native	Grossulariaceae
Ribes sp	NA	NA	Native	Grossulariaceae
Rosa acicularis	Prickly rose	Upland	Native	Rosaceae
Rubus idaeus	Common red raspberry	Upland	Native	Rosaceae
Rumex maritimus	Golden dock	Hydrophyte	Native	Polygonaceae
Rumex occidentalis	Western dock	Hydrophyte	Native	Polygonaceae
Rumex sp	NA	NA	NA	Polygonaceae
Salix petiolaris	Meadow willow	Hydrophyte	Native	Salicaceae
Salix sp	NA	Hydrophyte	Native	Salicaceae
Schoenoplectus tabernaemontani	Softstem bulrush	Hydrophyte	Native	Cyperaceae
Scirpus microcarpus	Red-tinge bulrush	Hydrophyte	Native	Cyperaceae
Scutellaria galericulata	Hooded skullcap	Hydrophyte	Native	Lamiaceae

Sisyrinchium montanum	Strict blue-eyed-grass	Hydrophyte	Native	Iridaceae
Sium suave	Hemlock water-parsnip	Hydrophyte	Native	Apiaceae
Solidago canadensis	Canadian goldenrod	Upland	Native	Asteraceae
Sonchus arvensis	Field sow thistle	Hydrophyte	non-Native	Asteraceae
Sparganium eurycarpum	Broad-fruit burr-reed	Hydrophyte	Native	Sparganiaceae
Sphenopholis intermedia	Slender wedgescale	Hydrophyte	Native	Poaceae
Stachys palustris	Marsh Hedge-Nettle	Hydrophyte	Native	Lamiaceae
Stellaria crassifolia	Fleshy starwort	Hydrophyte	Native	Caryophyllaceae
Symphoricarpos albus	Common snowberry	Upland	Native	Caprifoliaceae
Symphyotrichum boreale	Northern bog aster	Hydrophyte	Native	Asteraceae
Symphyotrichum ciliolatum	Lindley's aster	NA	Native	Asteraceae
Symphyotrichum puniceum	Purplestem aster	Hydrophyte	Native	Asteraceae
Tanacetum vulgare	Common tansy	Upland	non-Native	Asteraceae
Taraxacum officinale	Common dandelion	Upland	non-Native	Asteraceae
Tephroseris palustris	Marsh fleabane	Hydrophyte	Native	Asteraceae
Thalictrum venulosum	Veiny-leaf meadow-rue	Hydrophyte	Native	Ranunculaceae
Thlaspi arvense	Field pennycress	Upland	non-Native	Brassicaceae
Trifolium hybridum	Alsike clover	Upland	non-Native	Fabaceae
Trifolium repens	White clover	Upland	non-Native	Fabaceae
Triglochin palustris	Marsh arrow-grass	Hydrophyte	Native	Juncaginaceae
Tripleurospermum inodorum	Scentless mayweed	NA	non-Native	Asteraceae
Triticum sp	NA	NA	non-Native	Poaceae
Typha latifolia	Broadleaf cattail	Hydrophyte	Native	Typhaceae
Urtica dioica	Stinging nettle	Hydrophyte	Native	Urticaceae
Veronica peregrina	Neckweed	Hydrophyte	Native	Scrophulariaceae
Viburnum edule	Squashberry	Hydrophyte	Native	Caprifoliaceae
Vicia americana	American purple vetch	Upland	Native	Fabaceae

Viola canadensis	Canadian white violet	Upland	Native	Violaceae
Viola renifolia	Northern white Violet	Hydrophyte	Native	Violaceae
Unknown Juncaceae	NA	Hydrophyte	Native	Juncaceae
Unknown Lamiaceae	NA	NA	NA	Lamiaceae
Unknown Poaceae 1	NA	NA	NA	Poaceae
Unknown Poaceae 2	NA	NA	NA	Poaceae
Unknown Poaceae 3	NA	NA	NA	Poaceae
Unknown Poaceae 4	NA	NA	NA	Poaceae
Unknown Poaceae 5	NA	NA	NA	Poaceae
Unknown Poaceae 6	NA	NA	NA	Poaceae
Unknown Poaceae 7	NA	NA	NA	Poaceae
Unknown Poaceae 8	NA	NA	NA	Poaceae
Unknown Poaceae 9	NA	NA	NA	Poaceae
Unknown Poaceae 10	NA	NA	NA	Poaceae
Unknown Poaceae 11	NA	NA	NA	Poaceae
Unknown Poaceae 12	NA	NA	NA	Poaceae
Unknown Poaceae 13	NA	NA	NA	Poaceae
Unknown Poaceae 14	NA	NA	NA	Poaceae
Unknown Poaceae 15	NA	NA	NA	Poaceae
Unknown sp 1	NA	NA	NA	NA
Unknown sp 2	NA	NA	NA	NA
Unknown sp 3	NA	NA	NA	NA
Unknown sp 4	NA	NA	NA	NA
Unknown sp 5	NA	NA	NA	NA
Unknown sp 6	NA	NA	NA	NA
Unknown sp 7	NA	NA	NA	NA
Unknown sp 8	NA	NA	NA	NA

Unknown sp 9	NA	NA	NA	NA
Unknown sp 10	NA	NA	NA	NA
Unknown sp 11	NA	NA	NA	NA
Unknown sp 12	NA	NA	NA	NA

Appendix D. Stress plot for non-metric multidimensional scaling (NMDS) of vegetation community composition.



Curriculum Vitae

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Salaria, S., Beazley, K., & Bush, P. (2016). *Landscape connectivity for whom, what and where-to-where: Engaging experts in connectivity modeling in Halifax, Nova Scotia*. Oral Presentation. Atlantic Division of The Canadian Association of Geographers 27th Annual Meeting, May 30 - June 4, 2016, Halifax, Canada.