Vegetation Based Assessment of Wetland Condition in the Prairie Pothole Region

by

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Author's Declaration

I hereby declare that I am the sole author of this thesis. This is a true copy of the thesis, including any required final revisions, as accepted by my examiners.

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Abstract

The northern prairie pothole region (NPPR) in central and southern Alberta contains numerous shallow, open-water pothole wetlands that provide important ecosystem services to the region, such as flood mitigation. To address the ongoing destruction of these systems, the Alberta government has put forth a new wetland policy to mitigate wetland loss and mandate wetland restoration to offset wetland loss. However, to evaluate the success of wetland restoration a tool is needed to assess wetland condition. An ideal tool for this management objective is a multimetric index (MMI).

Multimetric ecological assessments such as the Index of Biotic Integrity use the responses of a specific biotic group as an indicator of disturbance, alleviating the need for complex direct measures of anthropogenic disturbance. Multimetric indices are used throughout the world to assess the condition of several ecosystems, and are applicable to wetlands. Wetlands in the NPPR have unique vegetation that is responsive to anthropogenic disturbance. I hypothesize that both the floristic attributes of wetland vegetation in this region and the distinct patterns of vegetation zonation could be used to produce a multimetric index that reliably indicates the condition of wetlands in agricultural areas. Seventy-two wetlands were sampled in central and southern Alberta in the summer of 2014 and 2015. Each wetland had its vegetation communities delineated and mapped using an SX Blue II GPS/GLONASS receiver to create spatial metrics. Vegetation quadrats were used to obtain floristic metrics related to percent cover, species richness and species traits. Using these metrics, I tested both the traditional method and random selection method of building an MMI.

I successfully developed and validated MMIs for wetlands in central and southern Alberta using both floristic and spatial attributes of wetland vegetation. In addition, I was able to demonstrate that a random metric selection method, which allows metrics with weak relationships to disturbance to be incorporated generated a more sensitive MMI than the traditional method, which includes only the metrics most strongly related to disturbance.

Both of these multimetric indices can be used to monitor wetland conditions and evaluate the success of wetland restoration projects in Alberta, directly addressing the needs of the government of Alberta to meet the conditions of their wetland policy. The spatial index is the first step in scaling towards a remote sensing approach in performing wetland assessments with an MMI. My work will assist wetland monitoring in Alberta and can be used to guide restoration practitioners in their efforts to create natural wetlands.

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1. General Introduction

Wetland ecosystems are found across the globe and play an important role in many different landscapes providing important habitat and key ecosystem functions and services (Fay et al., 2016; Russi et al., 2013). Unfortunately, human development has caused extensive wetland loss and the loss of associated ecosystem services with estimates of wetland loss ranging from 33% (Hu et al., 2017) to as high as 71% since the 1900's (Davidson, 2014). The introduction of the Ramsar convention on wetlands in 1971 marked a rise in concern over and management of wetlands and as of 2008, 44% of the 160 member countries had adopted a national wetland policy (Ramsar Convention Secretariat, 2010). A key component of wetland management is the ability to assess the health and function of wetlands in order to gauge the success of wetland restoration and to preserve sensitive wetlands (Russi et al., 2013). While many jurisdictions have enacted regulations pertaining to wetlands, some regions are still lacking a framework for wetland protection. One such region is Alberta, Canada, which is home to a number of different wetland types, including marshes commonly referred to as prairie pothole wetlands. Recently, the province of Alberta has created a new wetland policy (Government of Alberta, 2013) but they continue to lack a suitable tool to assess the condition of wetlands within the areas affected by the wetland policy.

1.1 The northern prairie pothole region and prairie pothole wetlands

The Northern prairie pothole region (NPPR) of North America passes through six American states and, in Canada, extends into Alberta, Saskatchewan and Manitoba (Seabloom & van der Valk, 2003a). It is home to a large number of shallow, open-water wetlands, which give this region its name and result from the rolling topography that characterizes this region. These wetlands are largely unconnected by surface waters. The result of this isolated hydrology is a variation in wetland permanence broken down into the following classes: ephemeral (class I), temporary (class II), seasonal (class III), semi-permanent (class IV) and permanent (class V) (Alberta Enviornment and Sustainable Research Development (ESRD), 2015; Stewart & Kantrud, 1971). Fed primarily by spring snowmelt, these wetlands will begin the summer inundated and gradually lose water to evapotranspiration throughout the summer (Hayashi et al., 2016) with only class IV and class V wetlands keeping water throughout the year.

1.2 Wetland services & wetland loss

Prairie Pothole wetlands provide many ecosystem services to the region, including water filtration, flood mitigation, groundwater recharge and carbon storage (Bartzen et al., 2010; Beyersbergen et al., 2004; Keddy, 2000). In addition to these services, prairie pothole wetlands are a biodiversity haven in the landscape, providing habitat for several unique plant (Kantrud et al., 1989) and invertebrate species (Mitsch & Gosselink, 2007) and breeding habitat for waterfowl (Beyersbergen et al., 2004; Seabloom & van der Valk, 2003a). While the ecosystem services provided by prairie pothole wetlands are valuable, many of these wetlands have been destroyed since human settlement, with estimates indicating as much as 70% of historic wetlands have already been lost (Bartzen et al., 2010; Dahl & Watmough, 2007; Serran & Creed, 2015). Wetland loss is primarily driven by agricultural expansion with wetlands being drained to repurpose the land for crop production (Bartzen et al., 2010; Paradeis et al., 2010). In addition to agriculture, continued urban and industrial expansion contribute to wetland loss in the NPPR (Bartzen et al., 2010; Paradeis et al., 2010; Serran & Creed, 2015). Wetlands are also affected by alterations to surrounding upland habitat and fragmentation of the surrounding landscape which can impair wetland function and degrade wetland condition (Paradeis et al., 2010; van Meter & Basu, 2015; Verhoeven et al., 2008).

1.3 Monitoring and assessment

The province of Alberta has experienced the same degree of wetland loss as the rest of the NPPR with an estimated two-thirds of historic wetlands lost, and contemporary losses continuing (Government of Alberta, 2013). To preserve wetlands and prevent additional loss of their valuable functions in Alberta, the Alberta Government instituted a new wetland policy in 2013 that aims to minimize wetland loss using a hierarchy of mitigation; 1) avoidance, 2) minimization and 3) replacement (Government of Alberta, 2013, 2016). If a wetland might be degraded by development or other human activity, the priority is to avoid degradation; if alteration cannot be avoided, the next step would be to minimize alteration to the wetland; if minimization is ineffective or impossible, replacement is mandated either through the creation of a new wetland or the restoration or enhancement of an existing wetland (Government of Alberta, 2013). This hierarchy recognizes that some wetland loss is unavoidable, and seeks to compensate for that loss through an offset program. Thus, wetlands removed from the landscape need to be replaced through restoration, enhancement, and creation or through the funding of research that contributes to improved wetland restoration (Government of Alberta, 2013, 2016).

Prior to the implementation of the new wetland policy, Alberta wetlands were managed under the interim wetland policy put into place in 1993 (Clare & Creed, 2014; Clare et al., 2011). The interim policy had a similar hierarchy which valued the avoidance and minimization of wetland impact over the removal and subsequent replacement of wetlands (Clare & Creed, 2014; Government of Alberta, 1993). However, Claire et al. (2011) found that all submissions to develop a wetland under the interim policy were approved, with replacement serving as the default option, bypassing the avoidance and minimization steps in the mitigation hierarchy (Clare et al., 2011). Under the new wetland policy, it is expected that wetland replacement will continue to dominate. Indeed, between 2014 and 2016 over 1700 submissions for wetland conversion were made to Alberta Environment and Parks, with no proposals being rejected (Matthew Wilson, Alberta Environment and Parks pers. comm.) Given the dependency of the policy on wetland restoration to achieve no net loss of wetland functions, ensuring the success of restored wetlands is integral. This requires monitoring and evaluation of restored wetland to ensure they are of adequate quality (Government of Alberta, 2016). Any monitoring would require an assessment method or tool that would be applicable across the settled area or "white zone" of Alberta, because this is the jurisdiction in which the policy has been implemented (Government of Alberta, 2013). This region spans a number of ecoregions and wetland types (Downing & Pettapiece, 2006; Government of Alberta, 2013).

1.4 Assessment tools: multimetric index

One of the most widely used ecosystem assessment tools is the index of biotic integrity (IBI) (Meador et al., 2008). The IBI is a multimetric index (MMI) that was first created by Karr (1981) to monitor the ecological condition of stream ecosystems by using the biotic response of fish communities to anthropogenic disturbance. Karr proposed the use of fish communities as an indicator of anthropogenic disturbance in stream habitats in Indiana, USA (Karr, 1981). Karr selected a number of metrics related to fish communities that he anticipated would change in response to stream disturbance, thereby providing a reliable estimate of the impact of human disturbance on a stream ecosystem (Karr, 1981; Karr & Chu, 2000). The metric values are standardized and combined to produce a single value which represents the integrated condition of the site (Barbour et al., 1995; Karr, 1981). The aim of Karr's IBI was to measure the biological integrity of rivers by analyzing the fish communities that use those rivers (Karr & Chu, 2000). An IBI provides an alternative method of environmental monitoring to exhaustive and frequently repeated measures of environmental characteristics because the response of the biotic community to disturbance is integrated over time (i.e., the current state of biota reflects the antecedent and current environmental conditions) and integrates across a number of distinct environmental stressors (i.e., the current state of biota reflects changes in a variety of water and soil quality parameters as well as changes to hydrology or other interacting populations; Karr, 1981; Karr & Chu, 1999; Schoolmaster et al., 2012).

Multimetric indices as a form of biomonitoring tool are beneficial because they use a number of metrics as opposed to a single bioindicator variable, which increases both the strength and reliability of the tool while improving detection capability (Barbour et al., 1995; Miranda et al., 2012; Karr & Chu, 1999). MMIs are also relatively easy to create and quite easy and affordable to use once developed (Karr & Chu, 2000). The results of MMI measurements are a single value for each site that represents the general integrity or condition of that site; this output is easy to interpret by entities that might be lacking sufficient ecological knowledge to understand usual laboratory output, but it can also be broken back down into metric scores that may help diagnose the cause of impairment or highlight necessary management actions to improve conditions (Karr, 1981; Karr & Chu, 2000). MMIs are also applicable to many different taxa including: fish (Karr, 1981), birds (Anderson, 2017; Wilson & Bayley, 2012), macroinvertebrates (Barbour et al., 1995), terrestrial vegetation (Mack, 2004; Wilson et al., 2013), and submersed aquatic vegetation (Rooney & Bayley, 2012b). In addition, the use of MMIs has grown from small-scale use to national and continental scales (Esselman et al., 2013; Schoolmaster et al., 2012; Stoddard et al., 2008). The application of MMIs has become so wide-spread that an estimated 90% of environmental assessments in the United States are done using an MMI (Barbour & Yoder, 2000).

Despite their popularity, MMIs face a variety of limitations and assumptions. First, since MMIs measure the response of a particular biotic group to disturbance, they are limited to systems that contain that biotic group. Further, there is an assumption that the selected biological group is representative of the whole ecosystem, including other unmeasured biological groups. The legitimacy of using some biological taxa as surrogates to infer the condition of other communities is commonly debated (e.g., Guareschi et al., 2015; Landeiro et al., 2012), though there is some evidence that commonly used bioindicators in wetlands in Alberta's NPPR are concordant (e.g., Rooney & Bayley, 2012a). Notably, MMIs based on biota are limited in terms of what time of year they can be applied, as field work is often constrained by the season in which biota are present and identifiable, which depends on the biological group being used. For example, plants must be sampled during flowering and fruiting to ensure they can be identified, whereas birds must be surveyed during the breeding season for auditory surveys to be effective. MMIs are typically based on the reference-degraded continuum approach, and are therefore constrained by the scope of the reference condition and the form of degradation used to develop and validate them. Thus, an MMI devised to evaluate marshes cannot be applied to a peatland, an MMI developed in South Dakota cannot be adopted in Alberta without additional validation, and an MMI designed to detect the effects of oil and gas activity will not necessarily detect the influence of agriculture, even on the same wetland type within the permissible region of inference. It is possible to create MMIs of broad scope (Mack, 2007; Miller et al., 2016), but that scope must be defined in advance and the resulting reference and degraded conditions adequately characterized. Presumably, a narrower scope of inference would permit the development of a more precise MMI by constraining the definition of the reference condition such that it is easier to detect the signal of wetland impairment against the noise of natural variability in wetland conditions. Indeed, Anderson (2017), when developing MMIs using the bird community in NPPR wetlands in Alberta, concluded that MMIs specific to the Grassland or the Parkland natural regions provided stronger relationships to disturbance than a single MMI that was devised to cover both regions. Nonetheless, Anderson (2017) was able to develop and validate a bird-based MMI that could apply to the Grassland and Parkland marshes in Alberta. She concluded that selection of MMI scope must balance the precision of the tool against the need for managers for a simple and universal tool that can apply to their entire jurisdiction.

1.5 Vegetation as an indicator

A multimetric index first needs a suitable biotic group to serve as an indicator of disturbance. Work has already been done to develop MMIs in the Alberta PPR using birds in the Parkland ecoregion (Wilson & Bayley, 2012) and across Alberta's entire PPR (Anderson, 2017). While invertebrates are a successful indicator in some regions, the hydrological differences between wetland permanence classes in the PPR makes them a poor choice (Gleason & Rooney, 2017). Likewise, submersed aquatic vegetation has been used in permanent wetlands in Alberta (Rooney & Bayley, 2012b), but temporary and seasonal wetlands do not retain ponded water long enough for submersed vegetation to be a consistently viable indicator. Wetland dependent, wet meadow vegetation has been successfully used as a bioindicator for permanent wetlands in Alberta's Parkland ecoregion (Wilson et al., 2013) so there exists the potential that vegetation could be used on a broad scale to assess wetlands across a range of permanence classes and across the entire PPR of Alberta.

Following a hierarchy of criteria for ecological indicators laid out by Dale and Beyeler (2001), ecological indicators should be 1) easily measured, 2) sensitive to stress, 3) respond to stress in a predictable manner, 4) an early signal of impending change in the system, 5) have a known response to disturbances, and 6) integrate across multiple stressors that could be associated with disturbance. Vegetation is relatively easy to sample as it is sessile, and requires no expensive equipment to identify and quantify it. Many studies have shown that wetland vegetation is sensitive to changes in hydrology (van der Valk, 1981), nutrient influx, and sedimentation (Gleason et al., 2003; van der Valk, 1981), at both the individual and community levels (Mitsch & Gosselink, 2007; Willby et al., 2001). As well, the response and predictability of wetland plants to stress has been well documented (Kantrud et al., 1989), hence the use of vegetation as an indicator in other marsh assessment tools (e.g., DeKeyser et al., 2003; Mack, 2007; Wilson et al., 2013). There are numerous wetland species that are adapted to a variety of conditions including variation in salinity (Euliss et al., 2004), and water level (van der Valk, 2005) meaning the presence of halophytes or drawdown-tolerant species could indicate a change to the wetland chemistry or hydrology, respectively. Finally, wetland vegetation can be integrated across multiple stressors; agriculture causes sedimentation, alterations to hydrology and nutrient influx (Bartzen et al., 2010), and vegetation integrates all of these environmental changes (Gleason et al., 2003; Mitsch & Gosselink, 2007; van der Valk, 1981). This response in vegetation is not only seen in reduced health of individuals, but also with changes at the community level (Keddy, 2000; Seabloom et al., 2001)

The dynamic hydrology of wetlands in the PPR results in a pattern of distinct vegetation zones or communities that change along a hydrologic gradient. This zonation results because hydrology is the primary environmental gradient that influences wetland vegetation, followed by salinity and disturbance (DeKeyser et al., 2003; Keddy, 1999; Seabloom & van der Valk, 2003b). These vegetation zones are consistent within wetland permanence class, but vary between classes providing a means to distinguish wetland permanence to the extent that vegetation zonation is used to classify wetlands on the basis of their permanence class (Alberta Enviornment and Sustainable Research Development (ESRD), 2015; Stewart & Kantrud, 1971). Wetland vegetation zones have distinct vegetation assemblages that vary from predominantly wetland obligates in wetter soils to a mix of facultative and upland species closer to the wetland edge (Stewart & Kantrud, 1971). Thus, the floristic composition of wetland vegetation and the spatial arrangement of vegetation zones are both sensitive to wetland hydrology and to any disturbances that might affect the wetland.

In part, but not exclusively, as a consequence of this dependence on hydrology, wetland vegetation communities and their resident species are highly responsive to disturbance. Agricultural activities in the surrounding watershed can result in nutrient contamination and sedimentation in wetlands (Gleason et al., 2003; Schindler & Donahue, 2006; Wright & Wimberly, 2013), along with physical alterations of wetlands from farming equipment, such as soil compaction (Taft et al., 1997).

Such activities might affect not only water quality in the wetland, but also water quantity (McCauley et al., 2015; van der Kamp et al., 2003). This is heightened by interannual climate variation which can result in a shift of wetland water depths from season to season. In a dry year, a wetland can become more accessible to farming machinery allowing agricultural activities to encroach further within the wetland boundary, resulting in physical disturbance such as plowing or compaction which would directly impair wetland function in wetter seasons when the wetland boundary returned to its previous extent.

1.6 Thesis objectives

The first goal of my thesis is to explore two different methods of creating a vegetation composition-based MMI for the assessment of ecological condition of wetlands in the Prairie Pothole Region of Alberta: the traditional method developed by Karr (1981) and refined by Stoddard et al., (2008) among others versus the iterative approach proposed by van Sickle (2010). The second aim is to develop and validate an MMI using the spatial arrangement of vegetation community patches as an indicator of agricultural disturbance, creating landscape metrics for vegetation community patches. In Chapter 2 of my thesis, I outline the procedures used to develop and validate an MMI using floristic vegetation metrics comparing two different methods for creating an MMI. In chapter 3, I used metrics derived from the spatial arrangement of vegetation communities as metrics to develop and validate an MMI. In chapter 4, I discuss the results of my data chapters and look at future work that could be carried out based on my results. My aim is to support wetland management throughout the PPR, but especially in Alberta, by enabling managers with robust and scientifically sound monitoring and evaluation tools to use in prairie pothole wetlands of varying permanence class.

2. Comparing methods of creating a multimetric index of wetland health using floristic vegetation metrics as indicators of disturbance

2.1 Introduction

Wetlands are an important natural global resource that are facing threats from expanding human activities. In the Prairie Pothole Region of North America, up to 70% of individual wetlands have been destroyed by human activities (Bartzen et al., 2010; Dahl & Watmough, 2007; Serran & Creed, 2015). This presents a growing problem as wetlands provide a number of essential ecosystem services including: carbon storage, groundwater recharge, nutrient cycling, and flood mitigation (Keddy, 2000; Mitsch & Gosselink, 2010). To offset the loss of ecosystem services when wetlands are removed from the landscape, regulators will often require the restoration or creation of new wetlands to achieve an end-goal of no net loss in ecosystem services (Brown & Lant, 1999; Government of Canada, 1991). To achieve these ambitious no-net-loss policy objectives, compensation wetlands must be of equivalent integrity and function as the natural wetlands whose loss they are intended to offset. As such, a successful restoration would be one in which the restored wetland possessed ecological and biological integrity (sensu Karr & Dudley, 1981) equivalent to natural wetlands in the region. Thus, evaluating restoration success necessitates a robust and reliable tool for the measurement of biological and ecological integrity (GoA, 2016; Kuehne et al., 2017). One of the most widely used assessment tools in North America is the multimetric index (MMI) which is used in 90% of environmental assessments in the United States (Barbour & Yoder, 2000).

2.1.1 Multimetric indices

Evaluating ecological integrity by measuring disturbance directly is challenging, as there are many forms of independent and cumulative disturbances that may affect an ecosystem (Schoolmaster et al., 2012). An MMI aims to characterize ecological integrity using indicators of disturbance, called metrics, to integrate the condition of an ecosystem over an ecologically meaningful timeframe (Karr and

Chu 1999). MMI's are favored over single-metric indicators, because they are composed of multiple measured metrics across a range of categories (e.g., diversity measures, abundance measures) that have a predictable response to disturbance (Hering et al., 2006). MMIs combine diverse metrics that have different responses to disturbance, combining the strengths of individual metrics to produce a single measure that has been shown to have a stronger response to disturbance (Barbour et al., 1995; Miller et al., 2016)

Metrics may be biotic or abiotic measurements of the ecosystem (e.g., Miller et al., 2016), though biotic metrics are most common. Measurements of water chemistry, hydrology and soil properties, though often sensitive and strongly related to environmental drivers, may be too spatially and temporally variable to provide a repeatable and accurate estimate of disturbance affecting an ecosystem (U.S.EPA, 2002). Biotic communities simplify the assessment as they can be sensitive to a wide variety of disturbance types and their responses may capture cumulative and synergistic effects that would be invisible if only individual drivers of disturbance or snap-shots of chemical condition were measured (Karr, 1981). Biological metrics may be categorized as reflecting measures of community structure (e.g., diversity, dominance), taxonomic composition (e.g., invasive species, sensitive taxa), individual condition (e.g., disease, contaminant levels), or biological processes (e.g., functional traits, productivity; Barbour et al., 1995)

To be selected, candidate metrics must respond sensitively and predictably to anthropogenic disturbance, but be relatively insensitive to natural environmental variation (Barbour et al., 1995; Karr & Chu, 1999; Hering et al. 2006). This is achieved following the reference condition approach (Bailey et al., 2014), whereby sensitivity is measured by sampling ecosystems across a gradient in disturbance from relatively pristine (i.e. reference) to highly degraded conditions (Stoddard et al., 2006). Due to rapid and extensive development of land by humans and the global influence of human activities, it is arguably impossible to find true reference sites for MMI development. However, a reasonable alternative is to sample sites that are in the "least disturbed condition," that most closely approximate high integrity, low disturbance conditions (Stoddard et al., 2006). Consequently, some independent (and preferably objective) estimate of disturbance is needed to build an MMI (e.g., Rooney & Bayley, 2010). Commonly, abiotic variables, measures of surrounding land use, or even best professional judgement are employed to rank sites on the basis of disturbance in MMI development (Hering et al., 2006).

Sampling across a disturbance gradient distinguishes multimetric from multivariate methods within the reference condition approach (Reynoldson et al., 1997), as multivariate tools are traditionally developed without defining the degraded condition. By sampling a large number of sites ($n \approx 50$) of both high and low ecological integrity, the multimetric method is able to separate natural variation in metric values observed among reference sites from variation between reference and degraded sites, which can be attributed to human disturbance (Hering et al., 2006).

MMIs are not universal: they are bound by a few key assumptions. First, an MMI is made for a certain ecosystem type; an MMI made for invertebrates in open water marshes could not be applied to rivers, as the diversity, species occurrence, abundance and functional traits of invertebrates occupying the two distinct ecosystem types will differ, and not because of any human influence (Karr, 1981). Second, sampling of the selected taxonomic group is assumed to be representative of the condition of the entire ecosystem and the other populations that occupy it. Finally, collection of field measurements should occur at the most suitable time to observe the characteristics of the taxonomic group being sampled. For example, if vegetation is the biotic basis of the MMI, then sampling must be carried out during the growing season, when species are identifiable.

2.1.2 Development in metric selection techniques

The first indices of biological integrity used best professional judgement to select metrics indicative of ecological condition (e.g., Karr 1981). Shortcomings resulting from the subjective and indefensible nature of relying on best professional judgement led to the development of more objective statistical approaches (e.g., Mack, 2004; Miranda et al., 2012; Rooney & Bayley, 2011; Stoddard et al., 2008). These approaches sought an optimal number of metrics that balanced the sensitivity and robustness of including multiple metrics against the redundancy and error propagation of including metrics that were collinear. The increased signal strength inherent in MMIs comes from including multiple independent measures of wetland condition, such that when summed together, the random sampling and measurement error in one metric cancels out the random sampling and measurement error in another metric, yielding a more robust and reliable index score overall (Schoolmaster et al., 2012). However, if the metrics included are simply derivations of the same root measurement, then it is likely that their errors will also be correlated and rather than cancel each other out, they will compound to destabilize the MMI (Stoddard et al., 2008; van Sickle, 2010). The amplification of error, or noise, would potentially obscure the signal of human disturbance (Figure 2-1), creating an MMI that produces erroneous results (Schoolmaster et al., 2012).

To prevent the inclusion of collinear metrics, MMIs traditionally employed a redundancy test and excluded metrics with correlation coefficients exceeding an arbitrarily selected threshold (e.g. Raab & Bayley, 2012; Rooney & Bayley, 2011; Stoddard et al., 2008). However, the approach creates a challenge in selecting which metric among the correlated metrics should be retained and which should be discarded, especially when both are equally sensitive to disturbance (Miranda et al., 2012).

More recently, MMI developers have recognized that correlated metrics themselves are not at issue. Rather, it is metrics with correlated error that destabilize MMIs. Indeed, since all metrics must be sensitive to the same measure of disturbance, some correlation among the metrics should be expected (Karr, 2006). Thus, some developers have based the redundancy test not on the metric values directly, but on correlation among the residuals from the relationships between each metric and disturbance (e.g., Anderson 2017, Schoolmaster et al. 2012). This enabled the inclusion of metrics that were highly correlated because they were all strongly related to disturbance, providing they had independent residual variation.

While this new approach avoided excluding useful metrics simply because they responded similarly to disturbance, it did not guarantee that the selected metrics collectively provided the strongest indication of disturbance out of all possible metric combinations, as only metrics that had strong relationships to disturbance individually were considered for MMI inclusion under this method. Even metrics that have weak relationships to disturbance could add strength to the overall MMI, so long as they explain variance in disturbance that is not accounted for by other metrics (Figure 2-1).

Van Sickle (2010) proposed a fundamentally different approach to metric selection; a random iterative approach. This novel method constitutes a data-driven strategy for MMI development. Rather than select the strongest independent indicators of disturbance and sum them, van Sickle (2010) advocated that we construct many MMIs using randomly selected metrics chosen from the pool of metrics that were individually sensitive to disturbance (not only the most sensitive metrics), then select from among the resulting whole MMIs rather than from among individual metrics (van Sickle, 2010). The method is unique in that it makes no attempt to control for redundancy among metrics by pre-screening what metrics are included in the randomly generated MMIs. The benefits of this method include increased objectivity as decisions about thresholds and metric inclusion are minimized. Further, by permitting the inclusion of less sensitive metrics, it is possible to devise an assemblage of metrics that is more strongly responsive to disturbance than by simply summing the most sensitive metrics (Figure 2-1).

In this chapter, I will develop and validate a multimetric index for the prairie pothole region of Alberta using vegetation as the indicator community. Vegetation is an ideal source of biological metrics for wetland evaluation because wetland plant community composition is strongly affected by anthropogenic disturbance (DeKeyser et al., 2003) and has been shown to be robust to inter-annual climate variation (Wilson et al., 2013). Further, vegetation-based MMIs have been successfully created elsewhere in Alberta (Raab & Bayley, 2012; Rooney & Bayley, 2011; Wilson et al., 2013). Adding to this toolset to expand its application across the NPPR has the advantage of smoothly integrating with existing and familiar evaluation practices.

Secondly, I will test van Sickel's (2010) assertion that the random generation of a large number of metric combinations can produce a stronger, more sensitive MMI (Method 2) than even a refined approach to redundancy elimination by pre-screening metrics for correlated error before selecting the metrics that, individually, are most sensitive to disturbance (Method 1). I will develop a vegetation-based MMI using both methods and compare the two to see which produces a more effective tool, validating them both against an independent dataset.

2.2 Methods

2.2.1 Study sites

I focused my work in the Grassland and Parkland Natural Regions of central and southern Alberta (Figure 2-2), which is home to numerous shallow, open-water marsh wetlands ranging in size and hydroperiod (AESRD, 2015). Alberta is currently implementing a new wetland policy and requires sensitive, reliable, scientifically-sound assessment tools to evaluate the success of wetland restoration projects in support of compliance monitoring (Government of Alberta, 2013). Within Alberta, the Grassland and Parkland Natural Regions are dominated by agricultural land use that bears responsibility for most historic wetland loss (Schindler & Donahue, 2006).

To capture the disturbance gradient from relatively pristine to highly degraded conditions, I selected 72 sites on the basis of the extent of non-natural land cover within 500 m buffers surrounding each wetland (more details in Anderson, 2017). Sites with 0-25% non-natural land cover, I considered low disturbance and I treated them as reference sites. Sites with 25-75% non-natural land cover, I considered medium disturbance, and I classified sites with 75-100% non-natural land cover as highly disturbed. Sites were selected to cover these three disturbance categories equally within the Parkland and Grassland Natural Regions.

Marshes were selected to ensure that they also spanned an independent gradient in hydroperiod, i.e., the duration of their ponded water (AESRD, 2015). Some sites are only briefly inundated, drying out by June (called temporarily-ponded). Others remain inundated through July (called seasonally-ponded). Whereas some contain ponded water throughout most of the growing season (May to September), except during droughts (called semi-permanently-ponded; AESRD, 2015). To ensure that any MMI I developed would apply to marshes of any hydroperiod, I selected the 72 wetlands such that each pond-permanence class was represented within each disturbance category.

2.2.3 Disturbance Scores

The categories of high, medium, and low disturbance that I used during site selection to ensure that I encompassed the entire gradient of disturbance necessary for MMI development were not sufficiently precise for evaluating MMI and metric sensitivity. The land cover data that I used in these determinations was sourced from the Agriculture and Agri-Food Canada Annual Crop Inventory (AAFC, 2013a, 2013b, 2013c, 2015) and though it has an overall reported classification error of < 15%, my ground-truthing revealed that it did not reliably discriminate between pastureland and native grassland. Anderson (2017) reported that 9 out of 10 sites in the Parkland region with 0% non-natural land cover, according to the AAFC crop inventory, had signs of cattle disturbance. Consequently, I based metric and MMI sensitivity measurement on disturbance scores that were modified from the percent of non-natural land cover within 500 m of each wetland. These non-natural percent cover values were modified by three factors: 1) the intensity of grazing activity evidenced in the wetland as observed by field technicians, 2) the presence or absence of pesticides in soil samples collected from each wetland and analyzed by Dr. Claudia Sheedy at the Agriculture Agri-Food Canada pesticide lab in Lethbridge, AB, and 3) whether agricultural activity crossed the actual wetland boundary, or whether grazing and cropping activity in the 500 m surrounding each wetland were separated from the wetland by a buffer strip. These additional values were non-continuous modifications that augmented the surrounding land-cover values based on field observations. These adjustments were adapted from Anderson (2017) and I provide more details describing them in Appendices 2.1 and 2.2. These disturbance scores ranged from 0 to 250.

2.2.4 Stratified random assignation of sites into development and testing sets

To validate the MMI, I required a dataset that was not used in its development (i.e., an independent test dataset). Most commonly, MMI developers achieve this by splitting the dataset into two portions and using one for MMI development while reserving the other for validation (Bailey et al., 2014; Lunde & Resh, 2012; Rooney & Bayley, 2011). I also needed to rarify my dataset to exclude pseudo-replication that would result from "double counting" the 24 wetlands that were sampled in both 2014 and 2015. To address this, I selected a random year using a random number generator with the selected year retained and the other year excluded from the analysis. The remaining sites were sorted by their disturbance score and sequentially assigned to the development dataset (66% of sites) or the validation dataset (33% of sites). This stratified random approach allowed me to assign sites to the development or validation sets randomly while ensuring that both sets had a representative distribution of low, medium, and high disturbance sites (Appendix 2.2).

2.2.5 Vegetation Sampling

I selected and sampled forty-eight sites in 2014 with 24 in the Parkland region and 24 in the grassland region. In 2015, I selected 24 sites sampled in 2014 to sample again in addition to 24 new sites, resulting in a total of 72 unique wetlands sampled. I sampled vegetation in late July/early August to coincide with the peak growing season, before plants senesce for winter. To obtain vegetation metrics at each wetland, I identified the vegetation communities, delineated and surveyed at an intensity proportionate to their size. I first identified and classified vegetation communities following the approach used by the Ontario Wetland Evaluation System (Ontario Ministry of Natural Resources, 2013). I based community classification on the combination of vegetation form and the identity of the dominant or co-dominant species. I delineated and mapped the vegetation communities using an SX Blue II GNSS receiver, which provided the area of the vegetation community. Vegetation was sampled based on the extent of the vegetation communities in each wetland; any community 100 m^2 or larger had a minimum of five 1 m^2 quadrats sampled, any community larger than 5000 m^2 had an additional 1 m^2 sampled for each 1000 m^2 above 5000 m^2.

I surveyed each quadrat using a modified Braun-Blanquet method to characterize ground cover (Appendix 2.3). I comprehensively characterized ground cover, including abiotic cover types (e.g., water, rock, litter, mud), as well as vegetation cover. I identified all plants to species where possible, following Moss and Packer (1983) with taxonomy updated by the Integrated Taxonomic Information System online database (http://www.itis.gov/; accessed January 2016).

2.2.6 Data preparation

I entered all ground cover data into a spreadsheet and performed quality assurance and control checks. I averaged the percent cover of plant species across all quadrats sampled at the wetland to produce a single percent cover value for each species at each wetland. I constructed a trait matrix to indicate the characteristic traits of each species of vegetation observed (Appendix 2.4). The relative abundance of a trait was then the sum of the percent cover of all species possessing that trait.

2.2.7 Metric Calculations

The process for metric calculation and MMI development for both methods follow the same steps for metric calculation and preparation, as well as metric standardization and scoring (Figure 2-3). Following Barbour et al. (1995), the initial pool of candidate metrics that I calculated included 735 measures of community structure, taxonomic composition, and biological processes (Appendix 2.5). I did not assess the health of individual plants. Metrics based on individual species or traits were calculated in three ways: 1) the percent cover of that species or trait; 2) the presence or absence of that species or trait; and 3) the proportion of total richness comprising that species or trait. I considered all three methods of metric quantification in developing the MMIs.

2.2.8 Metric pre-screening

The primary criterion for metric selection is that they must be sensitive to disturbance. To determine which metrics were sensitive to disturbance, I ran a Spearman correlation between metric values and disturbance score, in R 3.4.2 (R Core Team, 2017) using the Hmisc package (Harrell & Dupont, 2017). The non-parametric Spearman coefficient was preferable to the more commonly used Pearson correlation coefficient because it does not assume a linear relationship between my metrics and disturbance, which better reflects the non-linear nature of many ecological relationships (McCune & Grace, 2002). Under the traditional method only the metrics most strongly related to disturbance are selected for inclusion, whereas under van Sickle's (2010) approach to MMI development (Method 2) the entire philosophy is that even metrics with weak relationships to disturbance individually can contribute importantly to an MMI that collectively is strongly related to disturbance. Thus, in place of the traditional $\alpha = 0.05$ threshold in sensitivity testing, I treated any metric values reasonably ($p < 0.1$) related to disturbance as adequately sensitive for consideration in the MMI.

A secondary criterion for metric inclusion in the MMI is that the metrics must possess sufficient range in their response to disturbance that the metric provides reasonable resolution in indicating disturbance level. I performed a coarse test of the range of each metric sensitive to disturbance by calculating the difference between the 5^{th} and 95^{th} quantile for each metric. If a metric had a difference of 0 (i.e. no difference between the 5^{th} and 95^{th} quantile values) it was deemed to have insufficient range to be considered further.

2.2.9 MMI Development: Method 1

This method incorporates a redundancy test to reduce the number of candidate metrics from those passing range and sensitivity tests (Figure 2-3). Following recommendations by Schoolmaster et al. (2012), the residuals from the Spearman rank correlation tests between each metric's values and disturbance scores were recorded and the correlation among residuals from different metrics was assessed using the Spearman correlation test. If two metrics had residuals with a Spearman rho value > 0.6, the metric with the weaker Spearman rank correlation coefficient with disturbance scores was excluded. Following the redundancy test, the remaining metrics were compared in terms of their sensitivity to disturbance and the strongest, most sensitive indicators selected for inclusion in the MMI.

2.2.10 MMI Development: Method 2

Whereas method 1 includes all sensitive and non-redundant metrics, method 2 uses random selection without replacement of a predetermined number of metrics to generate a large number of MMIs that are then compared to determine which MMI is most sensitive to disturbance. Thus, the number of metrics included in the MMI by method 2 must be determined *a priori*. I chose to compare MMIs with four, six, and eight metrics based on work by van Sickle (2010) and Magee et al. (in press).

I generated 50,000 four-metric MMIs by randomly selecting four metrics from the pool of 88 pre-screened metric values, without replacement, using R. The number 50,000 was an arbitrary compromise between van Sickel's (2010) 1000 iterations and the 2,441,626 possible four-metric combinations of the 89 metrics in the pool. If the randomly generated MMI included metrics whose residuals after regressing on disturbance included a pair-wise Spearman correlation coefficient with rho > 0.9 or if the four metrics had an average pair-wise Spearman correlation coefficient among their residuals from the regressions on disturbance whereby rho >0.75, then the MMI was rejected due to the risk of compounding error. Then the whole process was repeated to produces 50,000 six-metric and 50,000 eight-metric MMIs, each screened for compound error.

2.2.11 MMI Selection

Of the MMIs out of the initial pool of 50,000 that passed the test for compound error, I selected the optimal four-metric, six-metric, and eight-metric MMIs using Akaike information criterion (AIC), a model competition framework (Symonds & Moussalli, 2011). I relied on AIC (corrected for small sample sizes AICc) from the general linear model in which the MMI scores "predict" (i.e., indicate) the disturbance scores. AICc values were calculated in R 3.4.2 (R Core Team, 2017) using the AICcmodavg package (Mazerolle, 2017). The optimal 4-metric, 6-metric, and 8-metric MMI had the highest AICc weight.

2.2.12 Metric Standardizing and Scoring

Before MMI validation, the metric values must be converted into standardized scores that can be summed together to yield the MMI score. I achieved this by first subtracting the 5th percentile dividing each metric value by the 95th percentile minus the 5th percentile of the range of metric values in the development set (equations 1 and 2). This removes the effect of data extremes while capturing the natural range of variation. If the metric was positively correlated with disturbance, standardization was done by using equation 1; if the metric was negatively correlated with disturbance, equation 2 was used.

Standardized metric values were capped at a maximum of 100 and a minimum of 0 meaning a score above 100 was changed to 100 and a score below 0 was changed to 0.

$$100 - \frac{Metric\ Value - 5th\ quantile}{95th\ quantile - 5th\ quantile}\ x\ 100$$

Equation 1.

$$\frac{Metric\ Value - 5th\ quantile}{95th\ quantile - 5th\ quantile}\ x\ 100$$

Equation 2.

2.2.13 MMI Validation

Once each site had a standardized MMI score, the MMI needed to be validated. I carried out MMI validation using the same technique for both method 1 and method 2; simple linear regression of the MMI scores from the validation dataset against their disturbance scores. If the p-value from these regressions was $p < 0.05$, I considered the MMI validated. For method 2 the optimal 4-metric, 6-metric and 8-metric MMIs were validated in this way; if more than one MMI had $p < 0.05$, those MMIs were compared using AICc and the MMI with the highest AICc weight was selected as the best performing MMI.

2.2.14 Method Comparison

Having constructed MMIs by two distinct methods, I wanted to see which method produced an MMI that better indicated ecological condition. To achieve this, I compared the AICc weights for the

MMI developed via method 1 against the best performing MMI developed by method 2, including all 72 sites (i.e., recombining the development and validation sub-sets of data).

2.2.15 Supplementary Tests

Since the MMIs being constructed span two ecoregions, multiple wetland permanence classes and use data sampled over multiple years, supplementary tests were carried out to see if these factors biased the MMI results. An analysis of variance was carried out to test for any significant difference between MMI scores at wetlands of different permanent class. As well, a two-sample t-test was carried out to test for significant differences in MMI scores between ecoregions and between sampling years.

2.3 Results

My vegetation sampling efforts yielded 732 potential metrics (Appendix 2.5). However, only 88 were reasonably correlated with disturbance scores (Spearman rho $|r_s| > 0.241$, $p < 0.1$; Appendix 2.5). This set of sensitive metrics was included in MMI development by both method 1 and method 2.

2.3.1 Method 1

Under method 1, I incorporated first the metric most strongly indicative of disturbance scores and then added the next most strongly indicative metric that passed the redundancy test meaning they had uncorrelated residuals from a Spearman rank test of sensitivity to disturbance. I iterated this procedure until all candidate metrics were considered. Only 4 of 88 metrics were thus incorporated into the MMI**.** These four metrics included: 1) the percent cover of *Scutellaria galericulata* (SCUGALER); 2) the percent cover of *Hordeum jubatum* (HORJUBAT): 3) the presence or absence of annuals (Annual PA); and 4) the proportion of total richness comprised of native annuals or biennials (Native AB PRch) (Table 2-1).

The MMI that I developed by method 1 was strongly and significantly indicative of disturbance scores, based on the development dataset (Figure 2-4; Simple Linear Regression: $y = -0.7688x + 316.84$;

$R^2 = 0.3584$, $F_{1,46} = 25.700$, $p < 0.0001$). It was also successfully validated on the independent set of 24 sites (Simple Linear Regression: $y = -0.7467x + 341.53$; $R^2 = 0.2518$, $F_{1,22} = 7.404$, $p = 0.012$).

2.3.2 Method 2

Of the 50,000 MMIs generated for each set of four-, six-, and eight-metric combinations, most failed to pass the redundancy test. Either the residuals from regressing metric values on disturbance scores had one or more pair-wise Spearman correlation coefficients above 0.9 or the pair-wise Spearman correlation among metric value residuals exceeded 0.75. Specifically, only 27% of the 4-metric MMIs, 0.1% of the 6-metric MMIs, and 0.004% of the 8-metric MMIs passed the compound error test. Following model competition, the optimum four-metric, six-metric, and eight-metric MMIs were identified using AIC (Table 2-2). Comparing the best models of each size, the four-metric MMI provided the strongest indication of disturbance scores (Table 2-3, 2-4; Figure 2-5).

2.3.3 Method Comparison

Using AICc to compare models constructed by method 1 and 2, the MMI from method 1 yielded an AICc 762.19 compared to the AICc value from method 2 which was 754.93 (Table 2-5). A linear regression between method 1 MMI scores and disturbance scores yielded an R^2 of 0.2982, F-statistic = 29.75, df = 70, $p < 0.0001$ (Table 2-5). A linear regression for method 2 MMI scores and disturbance scores yielded an $R^2 = 0.3655$, F-statistic = 40.33, df = 70, $p < 0.0001$ (Table 2-5).

2.3.4 Supplementary Tests

I plotted method 2 MMI scores versus disturbance scores to visually assess any potential compounding effects of region, permanence and sampling year (Figure 2-6). An analysis of variance between wetland permanence class and MMI score yielded non-significant results (Table 2-6). There was no significant difference between scores from sites sampled in 2014 and sites sampled in 2015 based on a two-sample t-test (Table 2-7). However, there was a significant difference between scores for sites in the parkland and Grassland ecoregions based on a two-sample t-test (Table 2-7).

2.4 Discussion

Wetlands serve an important role in the landscape, providing valuable ecosystem services (Keddy, 2000; Mitsch & Gosselink, 2010). The continuing loss of these ecosystems and their associated ecosystem services has prompted many regulators to enact legislation that aims to mitigate this loss (Bartzen et al., 2010; Dahl & Watmough, 2007; Kuehne et al., 2017). Inherent in the process of managing any ecosystem is the need to evaluate it's ecological condition, which is increasingly carried out using multimetric indices (MMIs) (Barbour & Yoder, 2000; Kuehne et al., 2017; Magee et al. in press).

My first goal in this thesis chapter was to create a multimetric index to assess the ecological integrity of Albertan prairie pothole wetlands of varying hydroperiod using vegetation as an indicator taxon. I consider vegetation an ideal indicator of anthropogenic disturbance in wetlands as plant communities are strongly responsive to changes in hydrology (Euliss et al., 2004), sedimentation rates (Gleason et al., 2003), and water quality (Euliss et al., 2004), and vegetation-based MMIs have been successfully developed for permanent shallow-open-water wetlands in Alberta before (Raab & Bayley, 2012; Rooney & Bayley, 2011; Wilson et al., 2013).

My second goal was to compare the traditional method of constructing MMIs by selecting and combining non-redundant metrics that individually possess strong relationships to disturbance (method 1) and van Sickle's (2010) proposed method of generating numerous MMIs through random combinations of reasonably sensitive metrics and selecting the strongest indicator of disturbance from that group of MMIs (method 2). Though computationally more intensive, this second method has the advantage of potentially identifying combinations of metrics that are superior indicators of wetland integrity. This method also addresses some of the lingering subjectivity that remains in the traditional method of building MMIs (Miranda et al., 2012). With method 1 excluding redundant metrics, choices are informed by statistics but still require best professional judgement when metric correlations are very similar. Using method 2, best professional judgement is largely removed. The benefit to this statistical approach is reduced barriers to constructing and validating MMIs. Also, once a tool is built using method 2, the coding framework remains to be used again if needed, allowing transparency and repeatability in tool development. If a new sampling effort is put forth, or new metrics are measured or calculated for the original sites, the entire iterative process could be repeated to update the monitoring tool.

Using method 1, I constructed a four-metric MMI that was strongly and significantly indicative of disturbance scores of wetlands in the development dataset and successfully validated on an independent suite of sites. The first metric related to *Hordeum jubatum* (foxtail barley), which was positively related to disturbance scores. *Hordeum jubatum* is a native grass that is found throughout the Parkland and Grassland regions that is considered a weed in this area (Bubar et al., 2000). The positive association between *Hordeum jubatum* and disturbance is likely due to the weedy, opportunistic behaviour of *Hordeum jubatum* and its observed tendency to grow in ditches and areas that were recently physically disturbed. One method of exploring a plant species tolerance to disturbance is to use coefficient of conservatism (CC) values. CC values are assigned to plant species within a region on a scale of 0 to 10 based on its tolerance to disturbance (Wilson et al., 2013). Coefficients of conservatism reflect the consensus opinion among expert botanists on the disturbance tolerance of a species with low values indicating disturbance-loving taxa (Taft et al., 1997). Indeed, the coefficient of conservatism assigned to *H. jubatum* is 0 in the Dakotas (NGPFQA, 2001), 2 in Montana (Pipp, 2015), and 1 in Parkland Alberta (Wilson et al., 2013).

The second two metrics are related to the lifecycle of plants and were also positively related to disturbance scores. The presence or absence of annuals metric (Annual_PA) and the proportion of total richness comprised of native annuals and biennials (Native ABPrch) were likely positively related to disturbance because frequent disturbance tends to exclude perennials, which come to dominate later during wetland succession (Odland & del Moral, 2002). Wetland vegetation communities in this region tend to be dominated by perennial, clonal grasses and sedges, unless the wetland is experiencing a drawdown event, in which case annual and biennial forbs will colonize the exposed soil (van der Valk, 2005; Welling et al., 2012). While drawdown events occur naturally (e.g., Euliss et al., 2004), wetland disturbance (e.g., heavy grazing, tilling or mowing) can alter wetland hydrology (e.g., McCauley et al., 2015) or directly expose the soil surface similarly to a natural drawdown (Garth van der Kamp & Hayashi, 2008), and provide an opportunity for annual and biennial forbs to take root. Further, of the 47 annual species observed in our sites, 49% (n = 23) of those species were considered to be exotic or invasive (Government of Alberta, 2015; Moss & Packer, 1983), and of the 23 invasive or exotic species, 78% (n = 18) are considered weeds (Bubar et al., 2000). Thus, at least the presence or absence of annual species likely reflects the incidence of exotic or weedy species that would be associated with human activity and is indicative of ecological impairment in these wetlands.

The final metric was the percent cover of *Scutellaria galericulata*, a native, perennial, wetland obligate member of the Lamiaceae (mint) family. *Scutellaria galericulata* was not found at any sites in the quartile with the highest disturbance (disturbance scores: 169.79 – 243.3) and 64% of observations of *Scutellaria galericulata* were at sites in the quartile with the lowest disturbance scores (2.8 - 87.57), revealing its sensitivity to agricultural activity. Where the coefficient of conservatism value assigned to *H. jubatum* was 2 or less, that assigned *S. galericulata* was 7 in the Dakotas (NGPFQA, 2001), 6 in Montana (Pipp, 2015), and 6 in Parkland Alberta (Wilson, Forrest, et al., 2013), indicating that it is commonly recognized by botanists as a relatively disturbance sensitive species. *Scutellaria galericulata* is the only metric in this MMI to have a negative association with disturbance.

The MMI produced by method 1 was built using a set of sites that spanned a gradient of anthropogenic disturbance and a gradient of hydroperiod from temporarily-ponded to semi-permanently-ponded. Successful validation of this MMI was done on an independent set of sites that spanned the same gradients of disturbance and hydroperiod. Validation of this MMI is significant as it confirms that a single, simple vegetation-based MMI can be created that indicates the level of anthropogenic disturbance affecting a wetland regardless of hydroperiod and across two distinct natural regions within Alberta. The new wetland policy for the province of Alberta manages wetlands in the Parkland and Grassland ecoregions as one region called the “white zone” so a single tool that can be applied to wetlands across this region is preferable in providing intra-jurisdictional consistency. Previous vegetation-based MMIs developed in Alberta were only applicable to a single natural region and to only permanent shallow-open-water wetlands (Raab & Bayley, 2012; Wilson, Bayley, et al., 2013). Implementation of such narrowly applicable tools would lead to a patchwork across the jurisdiction and additional complexity in policy implementation.

Unlike method 1 where all suitably sensitive and non-redundant metrics are included in the MMI and thus the total number of metrics is determined by the process of metric selection, method 2 requires that the number of metrics to include in the MMI be determined in advance. Because this decision is arbitrary, I chose to compare 4-, 6- and 8-metric MMIs to ascertain which number of metrics yielded the optimal balance of sensitivity and simplicity. These MMI sizes were selected to reflect the typical number of metrics included in wetland MMIs reported in the literature (e.g., Karr & Chu, 1999; Raab & Bayley, 2012; Rooney & Bayley, 2011; U.S.EPA, 2002b) and the practical limitations of computational speed when working with higher metric MMIs. Using method 2, I was able to produce a validated MMI for each metric size class, indicating the general robustness of vegetation-based MMIs as even 8-metric MMIs were not overfit to the development dataset. Although validated MMIs including 6 and 8 metrics could explain more of the variance in wetland disturbance scores, the additional metrics did not provide significantly greater predictive strength, and ultimately my model competition process identified that the best 4-metric MMI was superior.   The AICc values for the best and second best performing 4-metrics MMIs were fairly similar (493.01 and 493.85) compared with the differences between the best performing 6- and 8-metric MMIs (6-metric: 502.19 and 504.05; 8-metric: 506.07 and 509.98). This likely reflects the differences in potential metric combinations for each metric size class.

The four metrics in the method 2 MMI included: the presence or absence of annuals or biennials which has a positive association with disturbance, the presence or absence of *Juncus balticus* which has a positive association with disturbance, the presence or absence of *Petasites frigidus* which has a negative association with disturbance and the presence or absence of *Carex* spp. which has a negative association with disturbance. As discussed with method one, the inclusion of a metric associated with annuals and biennials is understandable given their life history type and association with early successional stages (Odland & del Moral, 2002) and the tendency for annuals and biennials to be weedy exotics and opportunistic species that will colonize wetlands that have been disturbed (Galatowitsch et al., 2000). Seeming to deviate from that trend, *Juncus balticus* is a native, non-weedy, facultative wetland, perennial rush that was ubiquitous in our study, but found more commonly in disturbed wetlands. In total, 50% of my *J. balticus* observations occurred in sites from the highest quartile of disturbance scores (disturbance scores: 165.95 - 221.68). The coefficient of conservatism value assigned to *Juncus balticus* was 5 in the Dakotas (NGPFQA, 2001), 3 in Montana (Pipp, 2015), and 3 in the Alberta Parkland (Wilson, Forrest, et al., 2013).

In contrast, both *Petasites frigidus* and *Carex* spp. metrics were negatively associated with disturbance. *Petasites frigidus* is a native, facultative wetland perennial that was only observed at low disturbance sites (disturbance score < 26.12). It had coefficient of conservatism of 10 in the Dakotas

(NGPFQA, 2001) and 8 in Montana (Pipp, 2015). These results actually draw into question the coefficient of conservatism that was assigned *P. frigidus* in Alberta's Parkland, where it is given only a4 (Wilson, Forrest, et al., 2013). Sedges in the *Carex* genus are highly indicative of natural wetlands in this region and 14 different species of *Carex* were observed in my study (Appendix 2.4). Though *Carex* was ubiquitous in my study, it was excluded from 50% of sites in the highest quartile of disturbance (disturbance score: 175.49-243.30), whereas the genus was present in 94% of sites in the lowest quartile of disturbance scores (2.8-95.96). Indeed, work on succession in wetlands has noted that clonal, perennial sedges *Carex atherodes* and *Carex aquatilis*, two species I commonly observed in low-disturbance wetlands, are late successional species (Moss & Packer, 1983). The coefficient of conservatism values are species-specific, but for the 14 species that I observed they range from 4-10 in the Dakotas (NGPFQA, 2001), 3-9 in Montana (Pipp, 2015), and 3-7 in Parkland Alberta (Wilson, Forrest, et al., 2013), indicating that some members of the genus are more tolerant of disturbance than others. It is therefore interesting that a metric which aggregates all members of the genus proved superior to metrics focusing on single *Carex* species.

Conspicuously absent from either tool are metric related to the coefficient of conservatism values which are metrics commonly found in many other vegetation-based MMIs (e.g., Mack, 2007; Raab & Bayley, 2012; Wilson, Bayley, et al., 2013; Magee et al., in press). A likely explanation for the absence of floristic quality or coefficient of conservatism metrics is the lack of a floristic quality index for the Grassland region of Alberta. While an FQI exists for the Parkland and Boreal regions of Alberta (Wilson, Forrest, et al., 2013), there has not been one developed for the Grassland region. For my work, Grassland species were assigned CC values based on the Dakota (NGPFQA, 2001), and Montana (Pipp, 2015), and Alberta Parkland (Wilson, Forrest, et al., 2013) FQI. Since we observe differences between CC values from different regions, it is likely that the CC values used in the Grassland region of Alberta may not reflect the fidelity of observed species to the Grassland region. The creation of an FQI for the Grassland region might improve this sensitivity of this metric.

The supplementary tests carried out on potential confounding factors serve to show how the MMIs ability to indicate disturbance is robust to these factors. The non-significant results of the two-sample t-test between 2014 and 2015 date implies that the floristic MMI is robust to inter-annual variation. Similarly, the non-significant result of the analysis of variance comparing MMI scores to wetland permanence class indicates that the MMI works equally well evaluating wetlands of any permanence class. A significant result was seen in the test to compare MMI scores between the Grassland and Parkland ecoregion. This indicates that MMI scores are slightly higher on average in the Parkland region than the grassland region. While the MMI was validated across both regions, this difference indicates that caution should be used when comparing scores between sites in different ecoregions. The likely cause for this difference is the inclusion of the metric associated with *Petasites frigidus*, a species that is rare in the Grassland ecoregion (Moss & Packer, 1983) and was only observed in the Parkland ecoregion. While there is a difference between regions in the MMI scores, there was no significant difference seen in disturbance scores (Table 2-7).   The policy mandate of the government of Alberta is that both the Grassland and Parkland natural regions be managed as a combined region (Government of Alberta, 2013). It is, however, possible that a tool built for each region individually would provide a stronger indicator of disturbance in that region. Evidence for this can be seen in some of the species-based metrics in the MMIs I built. For method 1, while *Hordeum jubatum* was observed at sites across both regions, *Scutellaria galericulata* has only one observation in the Grassland compared to 11 observations in the Parkland. Similarly, with method 2, *Carex* spp., and *Juncus balticus* were commonly found in both regions, but *Petasites frigidus* was only seen in the Parkland region. In the case of *S. galericulata* and *P. frigidus*, both of these species are rare in the Grassland, commonly occurring in the Parkland or other natural regions in Alberta (ABMI, 2017; Moss & Packer, 1983). These metrics are essentially selecting for sites in the Parkland and an MMI specific to the Grassland would replace these metrics with metrics that better explain disturbance within the Grassland as opposed to within the joint management area. However, given the needs of the Alberta government, the tools I have made provide an efficient and reliable means to assess wetland integrity across the whole prairie pothole region of Alberta, providing an urgently needed management tool for a region where wetland management is essential for the preservation of key ecosystem services (Government of Alberta, 2013).

Implementation of either of the MMIs I have constructed would be easy to carry out, requiring technicians with limited botanical skills to perform site assessments during the growing season. Interestingly, because percent cover estimates did not yield superior metrics to the simple presence or absence of key indicator species, the site assessments could be dramatically simplified relative to how I gathered the data necessary for my thesis. Another interesting observation is the number of species specific metrics in both tools, method 1 having two species specific metrics, method 2 having three species specific metrics. The only trait-based metrics included in both tools were related to the lifecycle of vegetation species.

2.5 Conclusion

With continuing wetland loss comes loss of the ecosystem services provided by these wetlands. To protect these services, and maintain a healthy ecosystem the Province of Alberta has implemented a new wetland policy to mitigate wetland loss. A policy, however, is only as effective as the tools used to implement it. My work provides a scientifically valid tool that assesses wetland condition across a broad region of Alberta where wetland loss is an ongoing issue. The MMIs I created provide managers with an easy method to assess wetlands and obtain reliable information about wetland condition. The MMI is robust to year-to-year variability in climate and applies equally to wetlands of any permanence class covered by the new policy (Government of Alberta, 2013).

My work also provides insight into the creation of future MMIs, highlighting the benefits of moving to a more computationally intensive approach using the random creation of potential MMIs and the inclusion of metrics even if they are not strong predictors of disturbance on their own. Expanding on van Sickle's (2010) method, I was able to create an MMI that out performed an MMI made following the traditional method. The creation of this tool provides a template for future users to follow this same procedure to create a scientifically validated tool that minimizes the role of best professional judgement in metric selection.

With the need for proper management of wetland habitat on the rise in Alberta, my work represents a step forward in the monitoring of wetland ecological health and the enforcement of wetland policy. Use of these MMIs to evaluate the success of wetland mitigation projects and to track their progress over time would provide wetland managers with a clear, scientifically valid assessments of their ecological condition. In addition, my work provides a framework for creating new MMIs in different areas, following a defensible and objective approach that maximizes MMI predictive strength and sensitivity with the minimum number of metrics, while ensuring that correlated metrics do not result in compounded error.

2.6 Figures

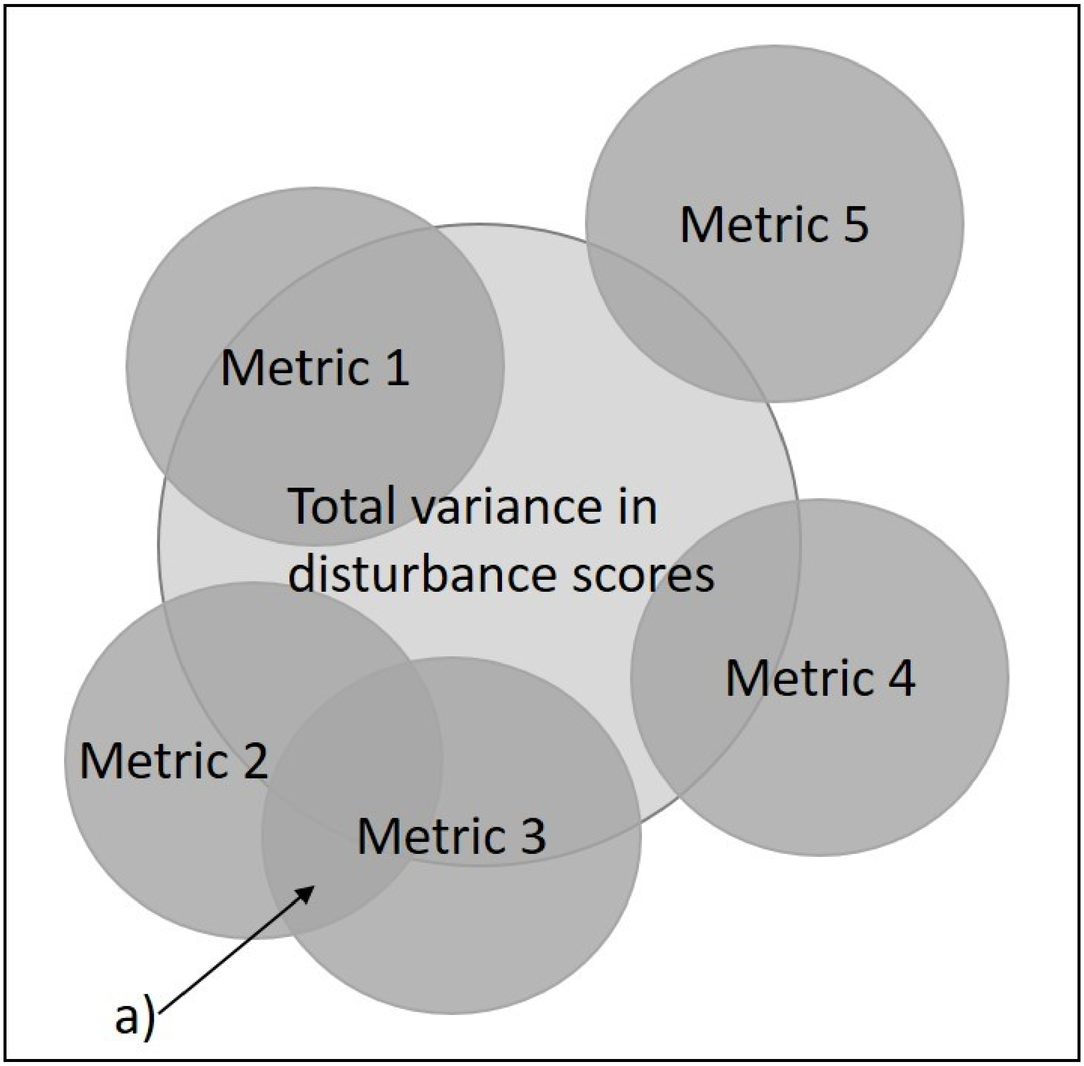

Figure 2-1: Visualization of how metrics in an MMI might combine to explain variance in disturbance scores. In this example, the proportion of variance in disturbance scores explained by each of the five metrics is represented by the degree to which they overlap the circle representing total variance in disturbance scores. The metrics range in their ability to indicate disturbance scores, with metric 1 having the strongest relationship to disturbance scores and metric 5 explaining the least, but nonetheless explaining a portion of otherwise unexplained variance in disturbance scores. Metric 5 thus illustrates the potential benefit to including metrics that explain a small amount of variance. Metric 2 and 3 represent metrics that are correlated, indicated by their overlap. A portion of the variance in disturbance scores that these two metrics explain is common, indicated by the overlap between metric 2 and 3 that occurs *within* the circle of disturbance score variance. The area indicated by a) constitutes an overlap *outside* the circle of disturbance score variance; this area represents a correlation in error variance that would produce compounded error if both metrics were included in the MMI.

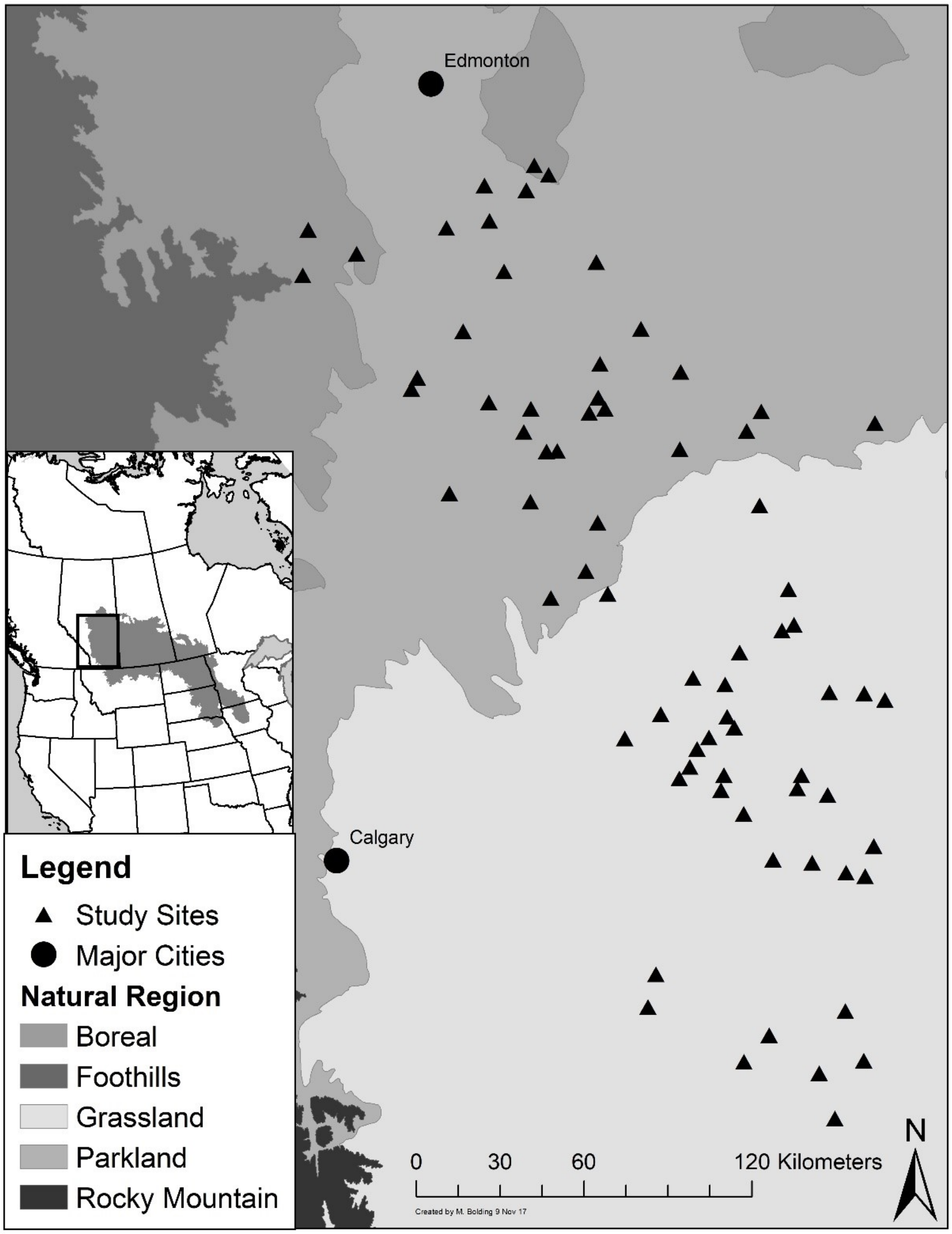

Figure 2-2: Map of study sites and their position within the ecoregions of Alberta.

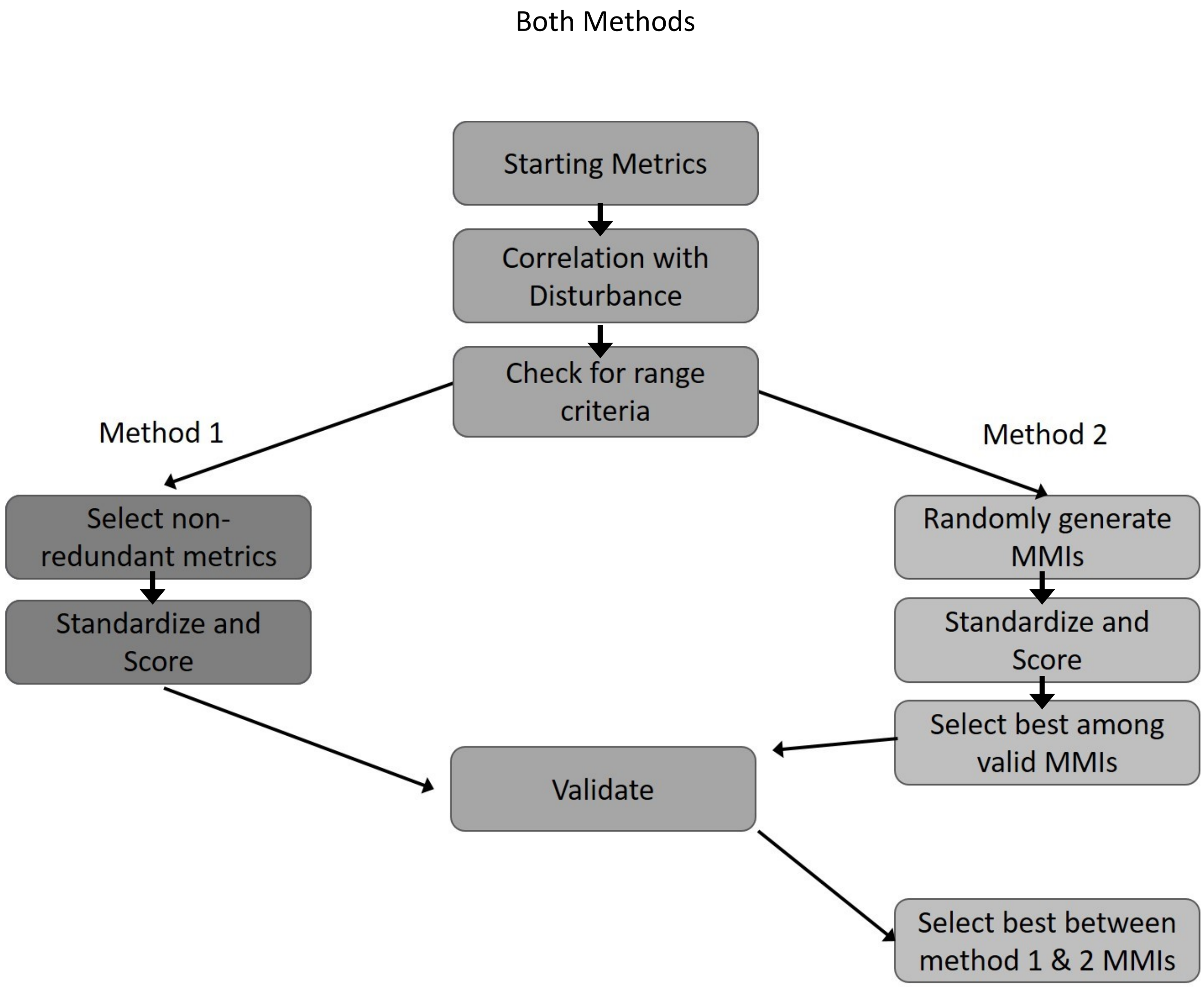

Figure 2-3: Breakdown of steps for constructing an MMI following method 1 and method 2. The steps that follow the same procedure for both methods are aligned in the center.

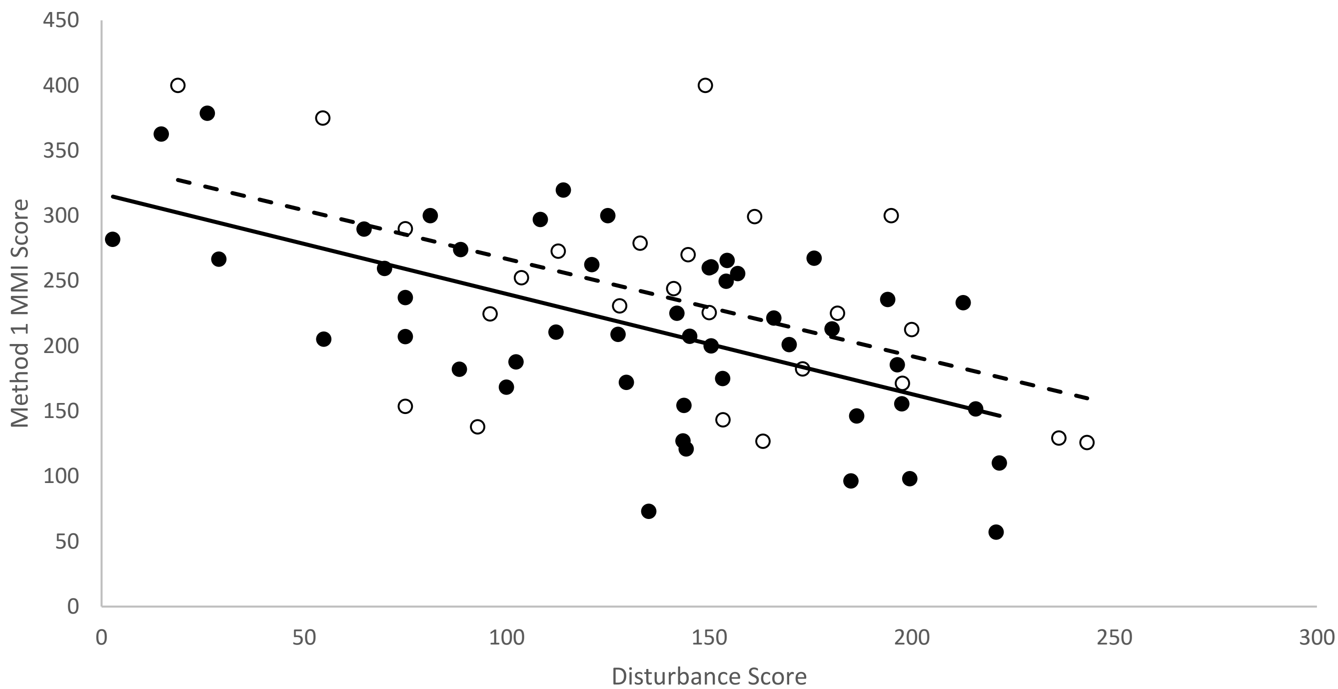

Figure 2-4: Linear regression of MMI scores versus disturbance scores from Method 1, for development sites (solid circles, solid line, n = 48, R^2 = 0.3584) and validation sites (open circles, dashed line, n = 24, R^2 = 0.2518). Method one metrics include: Percent cover of *Scutellaria galericulata*, percent cover of *Hordeum jubatum*, presence/absence of annuals, and the presence/absence of native annuals and biennials as a proportion of total species richness.

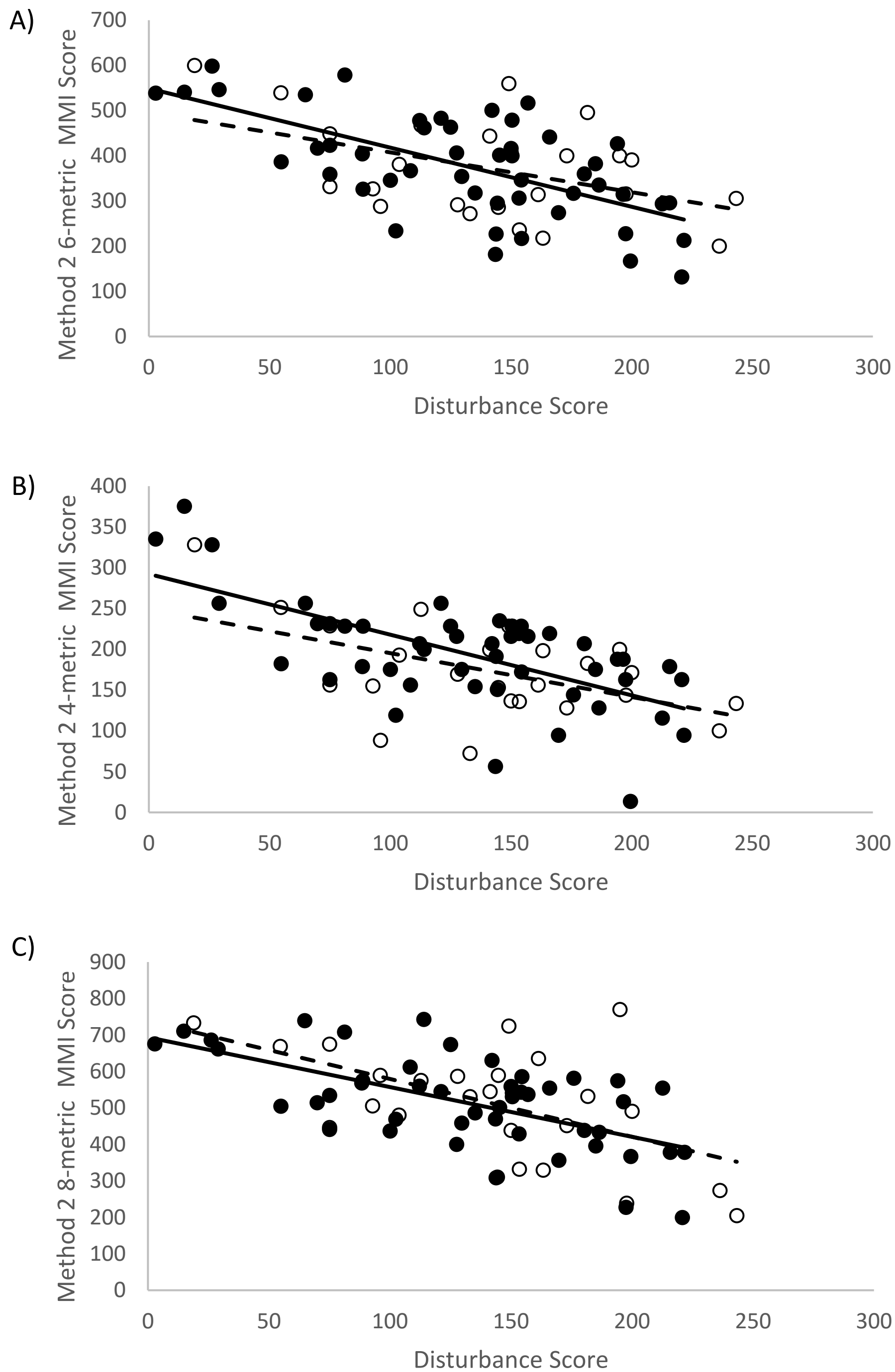

Figure 2-5: Linear regression of MMI scores versus disturbance scores from Method 2 MMIs showing development sites (solid circles, solid line, n = 48) and validation sites (open circles, dashed line, n = 24): A) Four-metric MMI, development R^2 = 0.4083, validation R^2 = 0.2706, B) Six-metric MMI, development R^2 = 0.4109, validation R^2 = 0.2032, C) Eight-metric MMI, development R^2 = 0.3613, validation R^2 = 0.3204. Note that the y-axis scale is not consistent among MMIs because the MMI scores are the sum of the metric scores, and thus the maximum score for an 8 metric MMI is double the maximum score for a 4 metric MMI. The MMI metrics are listed in table 2-4.

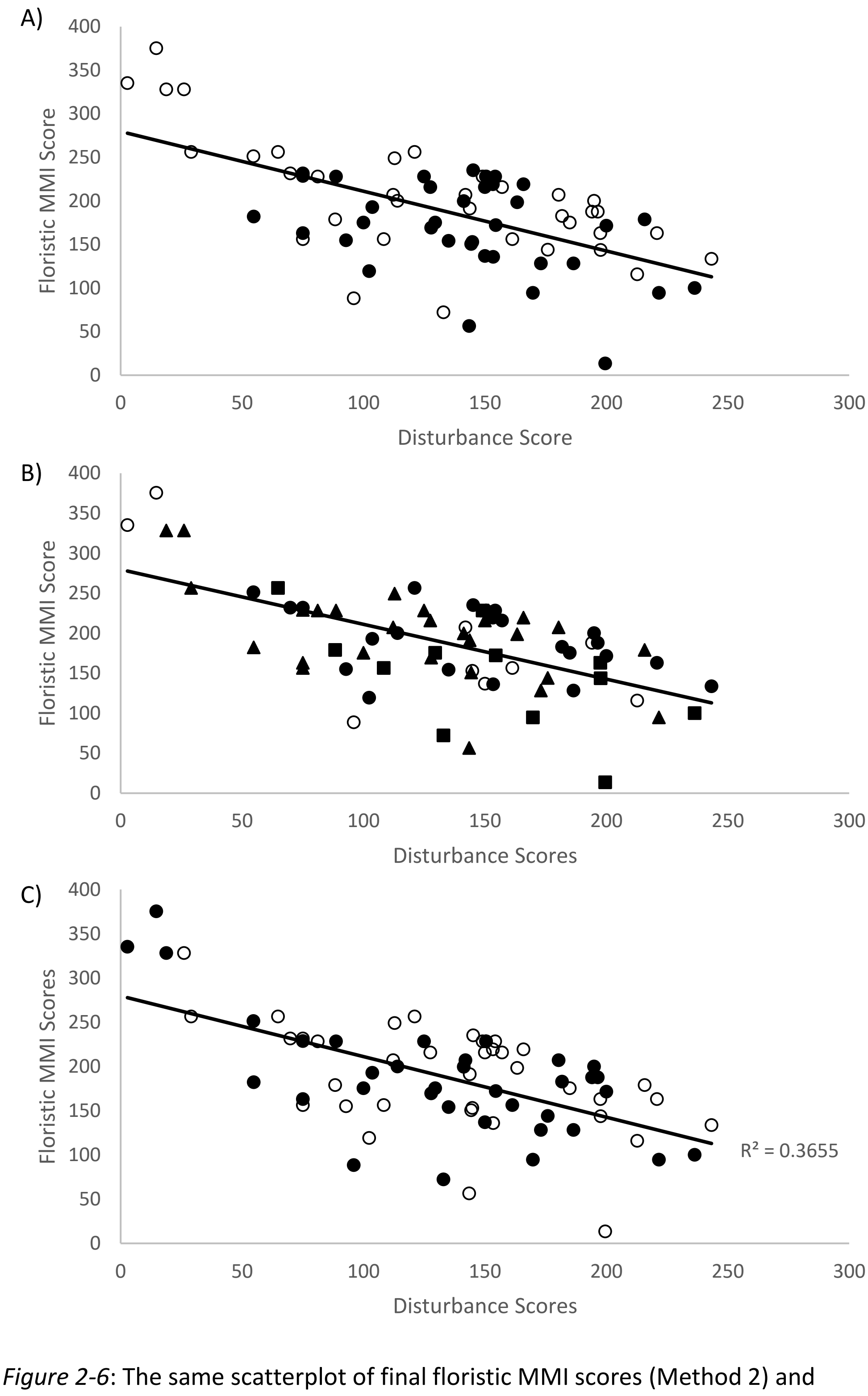

Figure 2-6: The same scatterplot of final floristic MMI scores (Method 2) and disturbance scores, but panels are coded by: A) region with the Grassland as solid circles and the Parkland as open circles, B) wetland permanence class with solid circles as temporary, triangles as seasonal, squares as semi-permanent and open circles as permanent wetlands, and C) sample year with 2014 sites as solid circles and 2015 sites as open circles.

2.7 Tables

Table 2-1: Metric details for MMI created using method 1.

Metric Names	Descriptions	Relationship with Disturbance	Standardization Formula	Spearman Rho	p-value
SCUGALER	Percent cover of *Scutellaria galericulata*	negative	$\frac{Metric\ Value - 5th\ percentile}{95th\ percentile - 5th\ percentile}\ x\ 100$	-0.5161	<0.0001
HORJUBAT	Percent cover of *Hordeum jubatum*	positive	$100 - \frac{Metric\ Value - 5th\ percentile}{95th\ percentile - 5th\ percentile}\ x\ 100$	0.4513	0.0012
Annual PA	Presence or absence of annuals	positive	$100 - \frac{Metric\ Value - 5th\ percentile}{95th\ percentile - 5th\ percentile}\ x\ 100$	0.2553	0.079
Native AB Prch	Proportion of total richness comprising native annuals or biennials	positive	$100 - \frac{Metric\ Value - 5th\ percentile}{95th\ percentile - 5th\ percentile}\ x\ 100$	0.5060	<0.0001

Table 2-2: The top MMI models in each category (4, 6, and 8 metrics) based on their AICc and their AICc Weights.

Number of Metrics	Model ID	AICc	Δ AICc	Model Likelihood	AICc Weights	Cumulative AICc Weights
4-Metric	MMI.9098	493.01	0.00	1.00	0.20	0.20
	MMI.7535	493.85	0.84	0.66	0.13	0.33
	MMI.583	494.33	1.32	0.52	0.10	0.43
	MMI.4599	494.44	1.42	0.49	0.10	0.53
	MMI.10342	494.66	1.65	0.44	0.09	0.62
	MMI.4477	494.73	1.72	0.42	0.08	0.70
	MMI.9504	494.79	1.78	0.41	0.08	0.78
	MMI.6926	494.90	1.89	0.39	0.08	0.86
	MMI.6000	495.09	2.08	0.35	0.07	0.93
	MMI.13448	495.10	2.09	0.35	0.07	1.00
6-Metric	MMI.48	502.19	0.00	1.00	0.35	0.35
	MMI.17	504.05	1.86	0.39	0.14	0.49
	MMI.23	504.43	2.24	0.33	0.12	0.61
	MMI.10	505.15	2.96	0.23	0.08	0.69
	MMI.1	505.27	3.08	0.21	0.08	0.77
	MMI.2	506.02	3.83	0.15	0.05	0.82
	MMI.7	506.10	3.91	0.14	0.05	0.87
	MMI.45	506.28	4.09	0.13	0.05	0.92
	MMI.46	506.46	4.27	0.12	0.04	0.96
	MMI.6	506.47	4.28	0.12	0.04	1.00
8-Metric	MMI.2	506.07	0.00	1.00	0.88	0.88
	MMI.1	509.98	3.91	0.14	0.12	1.00

Table 2-3: Parameters of final AICc comparison between the best performing 4-, 6-, and 8-metric MMIs created by Method 2, including all 72 wetlands.

	Parameter	4-Metric MMI	6-Metric MMI	8-Metric MMI
AIC comparison between MMIs	AICc	754.93	758.16	757.78
	Δ AICc	0.00	3.23	2.86
	Model Likelihood	1.00	0.20	0.23
	AICc Weights	0.70	0.14	0.17
	Cumulative AICc Weights	0.70	1	0.86
Linear regression of the development set	R^2 for Development Set	0.4083	0.4109	0.3613
	F Statistic	31.75	32.08	26.03
	p-value	< 0.0001	< 0.0001	< 0.0001
Linear regression of the validation set	R^2 for Validation Set	0.2706	0.2032	0.3204
	F Statistic	8.163	5.61	10.37
	p-value	0.009	0.0271	0.0039

Table 2-4: Method 2 MMI metrics and their properties.

Method 2 MMI	Metric Names	Descriptions	Relationship with Disturbance	Standardization Formula	Spearman Rho	p-value
4-Metric MMI	Carex PA	Presence or absence of the *Carex* genus	negative	$\frac{Metric\ Value - 5th\ percentile}{95th\ percentile - 5th\ percentile}\ x\ 100$	-0.0218	0.8832
	JUNBALTI PA	Presence or absence of *Juncus balticus*	positive	$100 - \frac{Metric\ Value - 5th\ percentile}{95th\ percentile - 5th\ percentile}\ x\ 100$	0.2518	0.0842
	PETFRIGI PA	Presence or absence of *Petasites frigidus*	negative	$\frac{Metric\ Value - 5th\ percentile}{95th\ percentile - 5th\ percentile}\ x\ 100$	-0.4194	0.003
	Annual Biennial PA	Presence or absence of annuals or biennials	positive	$100 - \frac{Metric\ Value - 5th\ percentile}{95th\ percentile - 5th\ percentile}\ x\ 100$	0.2416	0.0981
6-Mertic MMI	JUNBALTI PA	Presence or absence of *Juncus balticus*	positive	$100 - \frac{Metric\ Value - 5th\ percentile}{95th\ percentile - 5th\ percentile}\ x\ 100$	0.2518	0.0842
	Lamiaceae Prch	The proportion of total richness comprised of members of the *Lamiaceae* family	negative	$\frac{Metric\ Value - 5th\ percentile}{95th\ percentile - 5th\ percentile}\ x\ 100$	-0.2461	0.0918
	ALOAEQUA	Percent cover of *Alopecurus aequalis*	positive	$100 - \frac{Metric\ Value - 5th\ percentile}{95th\ percentile - 5th\ percentile}\ x\ 100$	0.2497	0.0869
	HORJUBAT PA	Presence or absence of *Hordeum jubatum*	positive	$100 - \frac{Metric\ Value - 5th\ percentile}{95th\ percentile - 5th\ percentile}\ x\ 100$	0.3743	0.0088
	Plantaginaceae Prch	The proportion of total richness comprised of members of the *Plantaginaceae* family	positive	$100 - \frac{Metric\ Value - 5th\ percentile}{95th\ percentile - 5th\ percentile}\ x\ 100$	0.2852	0.0494
	Annual Biennial Prch	The proportion of total richness comprised of annuals or biennials	positive	$100 - \frac{Metric\ Value - 5th\ percentile}{95th\ percentile - 5th\ percentile}\ x\ 100$	0.4703	0.0007

Method 2 MMI	Metric Names	Descriptions	Relationship with Disturbance	Standardization Formula	Spearman Rho	p-value
8-Metric MMI	Native annual biennial Prch	The proportion of total richness comprised of native annuals or biennials	positive	$100 - \frac{Metric\ Value - 5th\ percentile}{95th\ percentile - 5th\ percentile}\ x\ 100$	0.506	0.0002
	RUMFUEGI	Percent cover of *Rumex fueginus*	positive	$100 - \frac{Metric\ Value - 5th\ percentile}{95th\ percentile - 5th\ percentile}\ x\ 100$	0.2714	0.0644
	BECSYZIG PA	Presence or absence of *Beckmania syzigachne*	positive	$100 - \frac{Metric\ Value - 5th\ percentile}{95th\ percentile - 5th\ percentile}\ x\ 100$	0.2783	0.0555
	JUNBALTI	Percent cover of *Juncus balticus*	positive	$100 - \frac{Metric\ Value - 5th\ percentile}{95th\ percentile - 5th\ percentile}\ x\ 100$	0.2521	0.0839
	Carex	Percent cover of *Carex* spp.	negative	$\frac{Metric\ Value - 5th\ percentile}{95th\ percentile - 5th\ percentile}\ x\ 100$	-0.024	0.8714
	Zoochory	Percent cover of species whose primary distribution method is zoochory	positive	$100 - \frac{Metric\ Value - 5th\ percentile}{95th\ percentile - 5th\ percentile}\ x\ 100$	0.294	0.0425
	Annual Prch	Proportion of total richness comprised of annual	positive	$100 - \frac{Metric\ Value - 5th\ percentile}{95th\ percentile - 5th\ percentile}\ x\ 100$	0.5036	0.0003
	Native annual biennial	Percent cover of native annuals or biennials	positive	$100 - \frac{Metric\ Value - 5th\ percentile}{95th\ percentile - 5th\ percentile}\ x\ 100$	0.2992	0.0389

Table 2-5: Summary of AICc and linear regression parameters for final comparison between method 1 and method 2 MMIs.

MMI construction Method	AIC comparison between Method 1 and Method 2					Linear regression against disturbance for all sites (n = 72)			
	AICc	Δ AICc	Model Likelihood	AICc Weights	Cumulative AICc Weights	R^2	F-statistic	Degrees of Freedom	p-value
Method 2	754.93	0	1	0.97	0.97	0.3655	40.33	70	< 0.0001
Method 1	762.19	7.26	0.027	0.03	1	0.2982	29.75	70	< 0.0001

Table 2-6: Results of analysis of variance to detect any significant difference between floristic MMI scores among wetland permanence classes.

Source	Type III Sum of Squares	Degrees of Freedom	Mean Squares	F-Ratio	p-Value
Permanence Class	24,638.86	3	8,212.95	2.191	0.097
Error	254,851.04	68	3,747.81		

Table 2-7: Results of a two-sample t-test testing for a difference in floristic MMI scores or in disturbance scores between natural regions and between sampling years.

Floristic MMI	**t-statistic**	**Degrees of freedom**	**p-value**
Region	-2.55	70	0.013
Sampling year	-0.262	70	0.794
Disturbance Score			
Region	1.114	70	0.269
Sampling year	-0.312	70	0.756

3. Using the spatial arrangement of wetland vegetation communities as an indicator of wetland condition.

3.1 Introduction

The prairie pothole region (PPR) of North America is characterized by a rolling topography that produces shallow, open-water wetlands (van der Valk & Pederson, 2003). A defining characteristic of PPR wetlands is the patterns formed in wetland vegetation where different plant assemblages are found in distinct zones within the wetland (Stewart & Kantrud, 1971). The primary driver of this vegetation zonation is water depth, (Keddy, 2000; Seabloom et al., 2001), but is affected by other physical and chemical factors such as fertility, salinity, herbivory, and physical disturbance (Galatowitsch et al., 2000; Keddy, 1999). These factors lead to the formation of distinct vegetation assemblages, each associated with soil characteristics that are determined, primarily, by flooding (Keddy, 2000; van der Valk, 1981) These assemblages of vegetation possess unique growth forms, with floating and submerged aquatic vegetation occupying deep open water, emergent vegetation growing in shallower open water, wetland dependent and facultative forbs and grasses growing in wet soils, and upland species invading at the dryer margins of the wetland (Figure 3-1). This results from the progressive exclusion of plant species that lack adaptations that let them persist in more persistently flooded soils and competition release for species that possess such adaptations (Keddy, 1992; Shipley, 2010).

Prairie pothole wetlands have weak surface connections with other wetlands and they possess a variable hydrology that responds to the climate in the region (McCauley et al., 2015). These wetlands experience cycles in their water levels determined by precipitation (primarily winter snowfall) and evapotranspiration throughout the summer with some changes in response to large rainfall events (McCauley et al., 2015; Poiani & Johnson, 2012). As water level is the primary driver in wetland vegetation zonation, there is a corresponding fluctuation in the pattern of wetland vegetation zonation as water levels change (Keddy, 2000; Seabloom et al., 2001). As water levels drop, previously inundated sediment is exposed allowing annuals whose seeds have been dormant in the seedbank to germinate and colonize the exposed surface, while emergent species will die off due to low water levels (Poiani & Johnson, 2012). When flooding returns, the annuals will be drowned out, and emergent species will be able to re-establish in the wet conditions. While water level is the primary driver of these vegetation dynamics, there are other constraints including the composition and integrity of the seed bank, the dispersal ability of propagules, seedling germination characteristics, and species survivorship (Poiani & Johnson, 2012; Seabloom et al., 2001).

Moreover, recent work by Kraft et al. (in review) revealed a strong association between surrounding land cover and water and sediment quality in prairie pothole wetlands of Alberta. The conversion of land cover from natural grassland or forest to cropland creates an influx of both nutrients, which can cause eutrophication, and sediments which can impact invertebrate communities and submersed aquatic vegetation (Gleason et al., 2003; Kennedy & Mayer, 2002; Paradeis et al., 2010). Alterations to the surrounding topography and installation of roads can remove temporary connections that exist between wetlands; these connections serve to move vegetation propagules between the normally isolated wetlands (Galatowitsch & van der Valk, 1996; Seabloom et al., 2001). Upland disturbance also facilitates the introduction and propagation of invasive species into adjacent wetlands, which could alter the vegetation community composition and arrangement (Dekeyser et al., 2009). Most importantly, however, changes to land cover can alter wetland hydrology, for example increasing surface runoff by reducing soil infiltration rates (e.g., Van der Kamp et al. 2003) or by draining adjacent wetlands, resulting in consolidation of water in remaining wetlands (e.g., McCauley et al., 2015). Because surrounding land cover can influence water and sediment quality in the wetland (Kraft et al. in review), the flow of organisms in and out (Seabloom et al. 2001) and wetland hydrology (van der Kamp et al. 2003), vegetation zonation patterns are quite sensitive to surrounding land cover and land use.

Given the sensitivity of vegetation zonation to anthropogenic disturbance, I hypothesize that vegetation zonation could be used as an indicator of wetland ecological condition. I predict that wetland vegetation zones will not only be affected by agricultural activity (e.g., livestock grazing, cropping, haying) that takes place within the wetland boundary but also by activities and land conversion within the wetland's catchment. In other words, I expect that the spatial arrangement of vegetation zones will be influenced by the activities taking place within the wetland and in the surrounding landscape.

Landscape ecology aims to describe the structure, function and change of patches within a landscape (Leitao and Ahern, 2002; McGarigal & Marks, 1994). In this context, structure refers to the composition of patches and their spatial arrangement (i.e., configuration) , whereas function refers to the interactions among these elements (McGarigal and Marks, 1994). Though typically applied to landscapes comprising multiple ecosystems, the framework of landscape ecology can be used to explore the relationship between spatial patterns and ecological integrity within a single ecosystem. I used a landscape ecology approach, wherein vegetation assemblages of differing floristic composition are considered different "classes" and individual units of each assemblage are treated as "patches" within a wetland "landscape." I then tested whether vegetation zonation composition and configuration is responsive to the level of agriculture-related disturbance affecting a wetland and I will attempt to use measures of the shape, size and distribution of these vegetation zone "patches" as metrics to develop a multimetric index capable of evaluating wetland integrity. Despite how commonly they are used in aquatic ecosystem evaluations (e.g., Barbour and Yoder 2000), to the best of my knowledge no one has developed a multimetric index using composition and configuration metrics at the individual ecosystem scale.

In this chapter, I will use these composition and configuration metrics to develop and validate a multimetric assessment tool for the evaluation of prairie pothole wetland condition in the northern prairie pothole region of Alberta based on vegetation zonation patterns. I employed the same iterative approach to metric selection that I outlined in chapter 2.

3.2 Methods

3.2.1 Site selection

I selected 72 sites spread between the Grassland and Parkland natural regions of Alberta. All sampling at these sites took place between 2014 and 2015. The sites cover a range of permanence class from temporarily to permanently ponded water (sensu Alberta Enviornment and Sustainable Research Development (ESRD), 2015). Permanence was estimated using the Alberta Wetland Inventory and verified by on-site observations. Independent of permanence class, wetlands also spanned a gradient in the extent of anthropogenic disturbance. Disturbance was determined using the land cover within a 500 m buffer surrounding each site. Information about the percent of non-natural land cover was used to ensure that the selected wetlands adequately covered low-disturbance (0-25% non-natural land cover), medium-disturbance (25-75% non-natural cover), and high-disturbance (75% non-natural cover) bins. Data on land use was obtained from the Alberta Merged Wetland Inventory (AEP, 2014) and the Agriculture and Agri-Food Canada Annual Crop Inventory (AAFC, 2013a, 2013b, 2013c, 2015).

3.2.2 Disturbance scores

Disturbance scores were calculated for each wetland based on the surrounding land cover that was used during site selection. Following the procedure outline in chapter 2, the land cover estimates were augmented based on *in situ* observations of 1) grazing intensity within the wetland, 2) the presence of grazing or agriculture within the wetland boundary, and 3) the presence or absence of pesticides within the wetland based on laboratory analysis carried out by Dr. Claudia Sheedy at the Agriculture Agri-Food Canada pesticide lab in Lethbridge, AB. These three parameters were combined with the surrounding land cover to produce disturbance scores following the procedure outlined in chapter 2 and Appendix 2.1 and 2.2.

3.2.3 Vegetation mapping

To approach vegetation zonation from a landscape ecology perspective, the boundaries of each vegetation patch must be delineated and each patch must be categorized into specific vegetation assemblages. This was accomplished by mapping the boundary of each vegetation assemblage using an SX Blue II GPS/GLONASS receiver (Geneq Inc., Montreal, QC) and following a set of decision rules (Figure 3-3). Each wetland was visited between late July and early August to capture the peak vegetation growing season. Wetland communities were identified and delineated using a series of decision rules similar to those used in vegetation mapping by the Ontario Ministry of Natural Resources (sensu Ontario Ministry of Natural Resources, 2013). First, vegetation was classified by the dominant vegetation growth form (Appendix 3.3). Second, vegetation was classified by the dominant or co-dominant vegetation cover. In most cases this was a species (e.g., *Alisma triviale*) or genus (e.g., *Carex* spp.), but in some cases, the dominant cover was not vegetated (e.g., open water, bare ground).

Once vegetation assemblages were defined on the basis of their dominant growth form and cover type, their extent was mapped using an SX Blue II GPS/GLONASS receiver (Geneq Inc., Montreal, QC) while physically walking along the perimeter of each patch of each vegetation assemblage following a strict protocol (Figure 3-3). While walking, the GNSS receiver registered a point every 1 m, providing the position dilution of precision (PDOP) is less than 4, create polygons that were saved in a Juno T41 C handheld device (Trimble Inc., Sunnyvale, CA) using ArcPad 10.2 software (ESRI Inc., Redlands, CA). Since we were recording a point for every meter walked, patches that were less than 0.5 m^2 were not mapped and were aggregated with the assemblage that comprised the greatest proportion of the edge. The boundary of a wetland assemblage was delineated by changes in the vegetation growth form and changes in the dominant species. In cases where the patch boundary was ecotonal instead of sharp, I placed the boundary at the point where the adjacent assemblages were about equally present, i.e., in the middle of the "transition zone" between assemblages. This 50/50 rule is analogous to the decision rule used to define vegetation patches in the Ontario Wetland Evaluation System (Ontario Ministry of Natural Resources, 2013) and the 50% rule used by the US Army Corps of Engineers to identify if a plant community is hydrophytic (U.S. Army Corps of Engineers, 1987). The outer margin of the wetland was also mapped, following the 50% rule, such that any patches including < 50% relative cover of wetland obligate or facultative wetland plant species was excluded from the wetland.

3.2.4 Vegetation map processing

Vegetation assemblage patch polygons made by the SX Blue II receiver points were uploaded and audited in ArcMap 10.3 (ESRI Inc., Redlands, CA) to correct any topology errors that may be present in the data (Appendix 3.2).

Once all vegetation assemblage patch polygons were audited, I converted them from a vector format to a raster format using the Polygon to Raster tool in ArcMap 10.3 using a minimum cell size of 0.5 m to reflect the smallest patch measured during the mapping process. I then applied an 0.5 m wide upland edge border around each wetland using the Buffer and Merge tools. Lastly, I converted the raster map files for each wetland into TIFF format using the Raster to Other Formats tool.

3.2.5 FRAGSTATS and metric generation

I characterized vegetation zonation using a variety of composition and configuration metrics, which can be categorized into six classes, five of which (area/edge, shape, core area, contrast and aggregation) can be created at the patch, class, and landscape level, whereas the sixth category, diversity, can only be calculated at the landscape level (Appendix 3.1). A summary of these metric groups and their applicable levels can be found in table 3-1. These metrics provide a quantifiable description of the configuration characteristics of the vegetation communities within a landscape, in this case an individual wetland.

The program FRAGSTATS created by McGarigal and Marks (1994) calculates a number of configuration and composition metrics based on input data and GIS imagery at the scale of individual patches, classes, and whole landscapes. In my study, I used patch-level metrics from FRAGSTATS to compute metrics for individual patches of vegetation assemblages, class-level metrics to compute metrics for different vegetation assemblages, and landscape-level metrics to compute metrics characterizing the whole wetland. All TIFF files were uploaded to FRAGSTATS as well necessary supplementary tables such as class descriptors (a list that defines which raster cell value corresponds to which vegetation assemblage (Appendix 3.3)), a table defining edge depth and edge contrast. Edge depth was set at a fixed 1.5 m based on field observations. Thus, assemblages smaller than 1.5 m in width were considered entirely edge. The edge contrast table contains "weights" which indicate the edge contrast between two adjacent patches of different vegetation assemblage class ranging from 0 (no contrast) to 1 (maximum contrast) (McGarigal & Marks, 1994). In my study, I based the contrast weight on the difference in vegetation height, estimated from dominant vegetation growth form characterizing the vegetation assemblage class (Appendix 3.4).

3.2.6 Metric pre-screening

Metrics pertaining to individual patches within a vegetation assemblage were deemed too specific to be useful in wetland assessment, hence metrics were limited to the entire vegetation assemblage (vegetation assemblage-level, n = 23 metrics) and the entire wetland (wetland-level, n = 29 metrics; Appendix 3.5). Combining vegetation assemblage-level metrics and wetland-level metrics resulted in 1202 potential metrics to be evaluated for inclusion in the MMI (Appendix 3.5).

MMI creation followed a similar procedure to the iterative approach discussed in chapter 2 using metrics derived from the wetland's floristic composition (Figure 3-2). The 1202 metrics were first tested for suitable range: by measuring the difference between the 5th and 95th quantile for each metric value. If a metric had a difference of 0 it was deemed to have insufficient range to be considered further and was removed. Next, metrics were selected based on their sensitivity to disturbance scores: metrics that had a reasonable difference in value between the upper and lower quintile of sites ranked by their disturbance score were considered potential metrics. This sensitivity test was carried out as a Mann-Whitney U test on metric values at the low versus the high disturbance sites using the program R 3.4.2 (R Core Team, 2017) with an alpha value of 0.2. The Mann-Whitney U test was chosen as it does not require a normal distribution in metric values. Metrics that passed both the range and sensitivity tests were considered candidate metrics and included in the MMI development and validation procedure.

3.2.7 MMI creation

In total, I generated 150,000 potential MMIs including either 4, 6 or 8 metrics (50,000 MMIs each). MMIs were generated by randomly sampling metrics without replacement from the pool of candidate metrics. Of the 50,000 MMIs that were randomly generated for each MMI size class, those that violated a collinearity criteria were discarded. The criteria for collinearity was based on correlation among ranked residuals following a Spearman correlation that was carried out between all candidate metrics. MMIs that had a mean correlation below 0.7 and pairwise metric correlation below 0.9 were considered acceptable and considered as potential MMIs.

For MMIs that passed the collinearity criteria, the selected metrics were then standardized and scored using the method outlined in chapter 2. In brief, the 5th percentile value was subtracted from the metric value then was divided by the difference between the 5th and 95th percentile. If the metric value was positively correlated with disturbance, it was subtracted from 100. The results were multiplied by

100 with any values below 0 becoming zero and any values above 100 becoming 100. Metric scores were then summed across all metrics (4, 6, or 8) to produce an MMI score for each MMI.

I then compared MMIs within metric size classes using simple linear regression with MMI score as the response variable and disturbance score as the predictor variable, with n = 72 wetlands. Selection of the optimal MMI within each size class was based on AICc weights in R 3.4.2 (R Core Team, 2017) using the AICcmodavg package (Mazerolle, 2017). The MMI with the lowest AICc weight was selected as the best performing MMI within each metric size class.

3.2.8 MMI Validation

Since all my sampled wetlands (n = 72) were used to develop the MMIs, I cannot use the traditional leave-p-out cross validation technique for MMI validation. Instead, I devised a two-fold approach to MMI validation, considering the best performing (lowest AICc value) MMI within each metric size class and applying a combination of bootstrapping without replacement to assess any overfitting of our MMI scores to disturbance scores and a Mann-Whitney U test to assess the ability of our model to distinguish between the lowest and highest quintile of sites ranked by their disturbance scores. Each MMI that passes both the Mann-Whitney U test and the bootstrapping validation will be considered in the final MMI comparison.

First, I used bootstrapping without replacement to repeat the linear regression analysis between disturbance scores and MMI scores using a random subsample of 70% of my sites and recording the slope of the relationship between disturbance and MMI scores for that data subset. I repeated this 1,000 times and the resulting distribution of regression slopes was used to establish 90% confidence intervals by removing the 5th and 95th percentile values. If the 90% confidence interval around my slope value for a particular MMI were entirely below 0 (i.e., a negative slope, as expected from the hypothesized relationship between MMI scores which reflect ecological integrity and disturbance scores which reflect ecological impairment), then I could conclude that the slope is significantly negative and that my MMI were not overfitted to the disturbance scores of the 72 study wetlands used to develop the MMI.

My second validation technique was to use a Mann-Whitney U test, mirroring the approach used to test the sensitivity of individual metrics. This test involves separating the lowest and highest 20% of sites on the basis of their disturbance scores, and then conducting a Mann-Whitney U test in R (R Core Team, 2017) to evaluate whether MMI scores differ significantly between the upper and lower quintiles of disturbance scores.

The last step in MMI development and validation was to compare validated MMIs among the different MMI size classes to select the optimal MMI that best indicates wetland disturbance using the minimum number of metrics. I used an AICc model comparison framework, contrasting the optimal, validated MMI from each size class. AICc analysis was done in R 3.4.2 (R Core Team, 2017) using the AICcmodavg package (Mazerolle, 2017).

3.2.9 Supplementary Tests

To assess any potential bias in MMI scores attributable to ecoregion, wetland permanence class or sampling year, I plotted MMI scores against disturbance scores to visually assess any potential bias. Any apparent bias in the MMI would then be tested statistically with either a t-test or an ANOVA, followed-up with pair-wise Tukey's honestly significant difference tests.

The total area of my sample wetlands is varied and this could introduce a bias into an MMI based on landscape metrics, for example, whereby larger wetlands achieve higher scores not because of superior ecological integrity, but simply because they are large. I will test for this in my dataset by comparing the metrics in the best performing MMIs with area to see if they are biased by wetland area.

3.3 Results

My vegetation mapping at 72 wetlands yielded comprehensive maps comprising 106 different vegetation assemblages that I aggregated to produce 51 distinct vegetation assemblage classes and four non-natural classes (e.g., cropland, road, etc.; Appendix 3.3). Only vegetation assemblage classes were considered in metric calculation.

For each of the 51 vegetation assemblage classes, I calculated 23 class-level metrics and an additional 29 wetland-level metrics, yielding a total of 1202 metrics. Of these, 438 metrics passed the range test (Appendix 3.7). Of metrics with adequate range, 74 passed the sensitivity test, demonstrating a statistically significant difference in metric value between the 20th percentile lowest and highest disturbance sites (Appendix 3.6). Thus 74 candidate metrics were considered for inclusion in the randomly generated MMIs.

Following random generation of 50,000 4-metric, 6-metric, and 8-metric MMIs, only a subset passed the collinearity test. In total, 8966 4-metric MMIs, 285 6-metric MMIs, and one 8-metric MMI were accepted as non-redundant.

These accepted MMIs were evaluated in a model competition framework, using AICc to evaluate model performance (Table 3.2). As there was only one 8-metric MMI that had sufficient correlation it was automatically selected as the optimal MMI for its size class (Table 3.3).

Validation involved two steps. The results of bootstrapping to randomly sample 70% of sites without replacement produced a distribution of slope values for the regression of scores from optimal MMI of each metric size class on disturbance scores (Figure 3-4), but all were significantly below zero (90% CI was below zero based on 1000 iterations of bootstrapping; Table 3-4). The Mann-Whitney U test results reveal that the optimal MMIs were all able to differentiate the highest quintile of disturbance scores from the lowest quintile of disturbance scores (Figure 3-5, Table3-5).

In comparing the validated, optimal MMIs in terms of their AICc values, the 6-metric MMI was significantly better at indicating disturbance scores than either the 4-metric or the 8-metric MMI (Table 3-6).

Scatterplots comparing MMI scores to sampling year, ecoregion and wetland permanence class suggested that the MMI may provide lower scores to temporarily ponded wetlands, whereas more permanently ponded ones (i.e., classes 2-5) receive an equivalent range of MMI scores (Figure 3-6). A follow-up ANOVA test revealed that MMI scores did differ significantly among wetland permanence classes (Table 3-7). However, the Tukey's honestly-significant-difference test found no significant difference in MMI scores between any combination of permanence classes (Table 3-8). Two-sample t-tests between MMI scores and sampling year yielded non-significant results, as well a two-sample t-test between MMI scores and ecoregion.

Comparison of metrics with wetland area revealed that only one of the metrics included in the 4-, 6-, or 8-metric MMIs was significantly area-sensitive (Appendix 3.8): total edge of vegetation assemblages increased with wetland area.

3.4 Discussion

Efforts to conserve and restore wetlands in the northern prairie pothole region hinge on our ability to evaluate wetland condition (Government of Alberta, 2013). Evaluation tools are necessary to support regional monitoring of wetlands, to identify high quality wetlands in need of conservation or degraded wetlands in need of restoration. Further, under a wetland mitigation framework that permits natural wetlands to be destroyed, a scientifically validated wetland evaluation tool is critical to ensuring that replacement wetlands are of adequate quality. This is particularly important given the broad evidence that wetland restoration and reclamation rarely achieve wetlands of equivalent function to natural wetlands (e.g., Moreno-Mateos et al., 2012).

Wetland vegetation occurs in assemblages determined primarily by the hydrologic gradient within the wetland (Keddy, 2000; Seabloom et al., 2001; Stewart & Kantrud, 1971), but also by other factors including fertility (Galatowitsch et al., 2000; Keddy, 1999). Anthropogenic disturbance such as cropping and grazing livestock can lead to wetland degradation, altering vegetation community composition (Galatowitsch et al., 2000; McCauley et al., 2015; van der Kamp et al., 2003) and potentially changing the configuration of these vegetation zonation. I hypothesized that vegetation zonation would be responsive to disturbance associated with human activities in and surrounding wetlands, particularly those that influence wetland hydrology because the sensitivity of vegetation to hydrology is well established (Galatowitsch et al., 2000; Keddy, 1999; Paradeis et al., 2010) and indeed so consistent that vegetation is an essential diagnostic indicator in most North American wetland classification systems (e.g., ESRD, 2015; Cowardin et al., 1979; Stewart & Kantrud, 1971; Zoltai & Vitt, 1995). My goal in this chapter was to develop and validate a multimetric index (MMI) to evaluate the degree of agriculture-related disturbance at a wetland using the spatial arrangement of wetland vegetation communities.

The spatial approach to wetland assessment, specifically using spatially derived metrics to indicate disturbance, is a novel approach to assessing wetland condition and indicating anthropogenic disturbance. To the best of my knowledge this represents an innovation in wetland evaluation tool development, as the spatial arrangement of vegetation zones has never been used as the basis of a wetland assessment before. Given my success in developing and validating an MMI based on the arrangement of vegetation zones, I conclude that agricultural disturbance does have a significant effect on the composition and physical structure of vegetation assemblages. Namely, agricultural activities like cropping and grazing lead to increased sedimentation (Euliss et al., 2004; van der Valk, 1981), nutrient influx into wetlands (Houlahan et al., 2006), increased likelihood of invasive species introduction (Bartzen et al., 2010) and the removal of surface connections that may exist between nearby wetlands (Galatowitsch & van der Valk, 1996; Seabloom et al., 2001).

Although no one has previously used the spatial zonation of vegetation as the basis of a MMI, a number of vegetation-based multimetric indices have been previously developed for wetlands in the prairie pothole region (e.g., DeKeyser et al., 2003; Mack, 2004) including in Alberta (e.g., Raab & Bayley, 2012; Wilson et al., 2013, Chapter 2). Recently, an MMI has even been developed based on floristic composition that applies to all wetland types across the conterminous United States (Magee et al., 2017). A commonality among all these tools is that they require intensive sampling of wetland vegetation with identification of plants to the species-level. Several incorporate expert knowledge on the regional sensitivity or tolerance to disturbance in the form of coefficients of conservatism (e.g., Andreas et al., 2004; Wilson et al., 2013). These subjective coefficients of conservatism provide a numeric ranking for every species of plant present in the wetland, but such values are specific to each region and may be unreliable given changes in climate and resulting shifts in species distributions (e.g., Schneider, 2013). Even vegetation-based MMIs that exclude coefficient of conservatism values require identification of vegetation to the species level for the calculation of metrics (e.g., richness of native perennials (DeKeyser et al., 2003), richness of *Carex* species (Mack, 2007). In addition, several MMIs require labor intensive and time-consuming measures of plant biomass (e.g., DeKeyser et al., 2003; Raab & Bayley, 2012) or extensive knowledge of plant traits (e.g., native/exotic, monocot/dicot, annual/biennial/perennial; DeKeyser et al., 2003; Mack, 2007; Raab & Bayley, 2012; Wilson et al., 2013; Magee et al., 2017). Thus, MMIs are typically reserved for intensive assessments and are not considered suitable when a rapid assessment is required (U.S.EPA, 2017). The MMI that I developed and validated in this chapter eliminates the need for such finely resolved plant identifications or measures of plant productivity, and I suggest it could be considered a rapid assessment approach. My MMI requires only the field-identification of a few species or genera and requires no sampling or laboratory analyses. Further, because difficult to identify grasses, sedges and willows do not require species-level identifications, the period during which my spatial MMI can be rigorously applied in the field is extended compared with floristic composition MMIs, which can only be implemented during the window of the growing season when traits necessary for floristic identification are evident. There even exists the potential to adapt my spatial MMI to be applied via remote sensing, for example using high-resolution UAV imagery, to undertake rapid and remote assessments of wetland condition. Rapid, remote assessments are important tools in wetland policy implementation given the high density of wetlands in the PPR and limited resources for intensive site evaluations (U.S.EPA, 2017).

The advantages of eliminating species-level identifications and the potential advantages of an MMI that could be assessed from high resolution imagery are somewhat offset by reduced precision in wetland evaluation. A comparison between the floristic-based MMI and the spatial-based MMI showed that the floristic based MMI scores had a higher R^2 when compered to disturbance scores than the spatial based MMI (see Chapter 4). The two MMIs I developed in this thesis could be used in tandem, with the rapid spatial MMI offering a first approximation of wetland integrity and the floristic composition MMI providing additional precision in uncertain or, high stakes, cases.

While the spatial approach to wetland assessment might lack the resolution that normally comes from an intensive floristic composition-based wetland assessment, the spatial approach is perhaps most relevant from a wildlife perspective. This is because it directly incorporates the coarse habitat structure characteristics that are used by wetland birds when selecting sites. Much work has been done to show that wetland birds select wetlands based on landscape-scale metrics, such as vegetation form, patch size or complexity (Naugle et al., 1999; Poiani & Johnson, 2012; Puchniak Begley et al., 2012; Riffell et al., 2003; Riffell et al., 2001). Thus, this spatial assessment approach considers the same landscape metrics that birds use when selecting sites to nest and breed at, which likely aligns with the management priorities of many wetland restoration agents.

My initial pool of metrics contained 23 metrics associated with each cover class (n = 51) plus 29 additional metrics for the entire wetland as a whole for a total of 1202 starting metrics. After the range and disturbance relationship tests, the number of metrics dropped to 74 with only five of the starting 29 wetland-level metrics remained: two Area & Edge metrics (total edge and edge density), one core area metric (Number of disjunct core areas) and two aggregation metrics (aggregation index and landscape shape index). Interestingly, only three of the initial 51 cover classes passed both range and sensitivity tests: obligate *Carex* spp., *Salix* spp., and *Alisma triviale*. These three vegetation communities retained all 23 of their starting metrics. Below I discuss these cover classes and why they might be strong indicators of agricultural disturbance.

The obligate *Carex* cover class was an aggregation of all vegetation assemblages observed to be dominated by *Carex atherodes, Carex pelli, Carex retrosoa* and *Carex utriculata*. These four species were the only obligate *Carex* found to dominate a vegetation assemblage and they were aggregated based on their shared morphology and habitat needs (Moss & Packer, 1983; U.S.DA, 2017b). This simplified field-level identification significantly, as the species in this genus are commonly differentiated by close examination of the flowers (Moss & Packer, 1983), which are evident only during certain times of the year and not all individuals in a population of perennial *Carex* sp. will flower in a given summer. More, some evidence from Minnesota wetlands suggests that the guild level might be more indicative of land cover and land use changes than species composition (Galatowitsch et al., 2000). The *Carex* genus contains many species categorized as wetland obligate or facultative wetland species and is an important indicator of wetland type and permanence class (U.S.DA, 2017b). Obligate *Carex* spp. typically occur in the wet meadow zone of prairie pothole wetlands (Stewart & Kantrud, 1972), and several studies have identified the sensitivity of the wet meadow zone to disturbance (e.g., Raab &

Bayley, 2012; Wilsonet al., 2013). Sedges and other perennial species were found to be replaced with annuals or introduced perennials in recently cultivated wetlands in Minnesota (Galatowitsch et al., 2000), potentially due to the sensitivity of this assemblage to sedimentation (Werner & Zedler, 2002). The members of this genus that tended to dominate assemblages are all tussock-forming keystone modifiers that actually enhance floristic diversity by creating microtopographical heterogeneity (Werner & Zedler, 2002), thus metrics based on the abundance and distribution of *Carex* spp. obligates are ecologically meaningful.

Willows (*Salix* spp.) were found at just over 7% of sites (n = 17) with 13 of those sites being in the Parkland ecoregion. *Salix* spp. are shrubby plants that are often found in wet habitats and require a moist environment for seed germination (Argus, 2008; Kuzovkina & Quigley, 2005). The fluctuating hydrology of prairie pothole wetlands is ideal for the colonization of *Salix* spp. with germination occurring in drawdown phases when water levels are low enough for *Salix* to establish but high enough to prevent *Salix* from excluding other wetland vegetation (Timoney & Argus, 2006). Most species of *Salix* observed in our study were considered wetland obligates with the remainder considered facultative wetland species (U.S.DA, 2017b). Since willows will colonize during drawdown or desiccation events, willows are considered to be “disturbance adapted” species (Timoney & Argus, 2006), though total removal of willows was observed at wetlands affected by agriculture. Willow stands provide important nesting habitat for various birds (Chastant et al., 2017; Olechnowski & Debinski, 2008)

The northern water plantain, or *Alisma triviale* is a native perennial forb that was found in both the Grassland and Parkland natural regions of Alberta (Government of Alberta, 2015; Moss & Packer, 1983). The coefficient of conservatism values for *A. triviale* are 6 for the Grassland region and 4 for the Parkland region (NGPFQA, 2001; Pipp, 2015; Wilson et al., 2013). In the Parkland region, *A. triviale* was exclusively found at wetlands that were surrounded by row crops (n = 2). The sole occurrence in the Grassland was at a site that was heavily affected by drawdown the year it was sampled. *Alisma triviale* was characterized by Stewart and Kantrud (1972) as a drawdown species that was commonly found in tilled wetlands and persisted along a gradient of salinity ranging from fresh to moderately brackish (Stewart & Kantrud, 1972). Further, a study by Gleason et al. (2003) found that while sedimentation associated with disturbance tends to decrease the emergence success of propagules in the seed bank, *Alisma triviale* was able to germinate despite increased sedimentation (Gleason et al., 2003). This trait, perhaps, explains why *A. triviale* communities were only primarily found at sites with high agricultural disturbance.

I was able to construct MMIs including 4, 6, and 8 metrics that met criteria for range, sensitivity, and non-redundancy. These MMIs were not overly sensitive to the individual sites included in their development, as evidenced by a bootstrapping validation test that determined the slopes between MMI scores and disturbance scores were significantly negative. Further, these MMIs were able to readily distinguish between high and low disturbance sites, as evidenced by significant difference in MMI score between the top and bottom quintiles of disturbance scores. Though MMIs validated from each metric size class, I conclude that the 6-metric MMI provided the optimal number of metrics based on AICc values. While I recommend the application of the optimal 6-metric MMI for evaluating the integrity of wetlands exposed to agricultural disturbance in the Parkland and Grassland natural regions of Alberta. It is important to note that the 4-, and 6-metric MMIs had similar AIC scores and some examination of each tool was required before making a final decision.

The best performing 4-, 6- and 8-metric MMIs had a number of similarities. There were two metrics that were included in the final tools in each metric class: the aggregation index for the entire wetland, and the splitting index for wetland obligate *Carex* spp. This clearly shows the importance of these two metrics in these MMIs. While most of the metrics in these tools are class-based, each final tool has at least one landscape-based metric. Finally, aggregation metrics were also important for these tools with at least half of the metrics in the 4-, 6-, and 8-metric MMIs being aggregation metrics, including the two metrics found in all three MMIs. While there are a lot of similarities, I feel that the 6-metric MMI stands apart, not only because of the higher AIC score, but it incorporates a third metric category into the tools having two area/edge metrics, three aggregation metrics, and one contrast metric compared to the 4-metric MMI which only has area/edge and aggregation metrics.

This 6-metric MMI included metrics from the Area/Edge, contrast, and aggregation categories of landscape metrics with four metrics chosen at the vegetation-assemblage scale and two at the wetland scale. Since each metric is calculated using the same basic components (i.e., patch area, edge length and inter-patch distance) there is some correlation expected between landscape metrics (Hargis et al., 1998). This correlation, however, is controlled for in the iterative MMI generation process as MMIs with strong pairwise and mean metric correlation are removed from consideration. The resulting potential MMIs show a strong separation of metrics by both category and cover class (Table 3.2). Of the 10 MMIs considered for 6-metric MMI class, all but one MMI had the following metric configuration: 2 obligate *Carex* spp. metrics, 1 *Salix* spp. metric, 1 *Alisma triviale* metric and 2 landscape metrics. Multiple metrics from within the same cover class (e.g., *Carex* spp.) were almost always from different metric categories with the exception of the landscape aggregation index and landscape shape index which, while both aggregation metrics, are calculated in different ways (Appendix 3.1). The AICc values for the top 10 6-metric MMIs are not strongly dissimilar, meaning the improvement between the most likely model and the second most likely is small. The benefit of using the iterative approach to MMI creation is that the final model is statistically better than the others as random selection of metrics can provide a better combination of metrics than simply selecting the metric that is most indicative of disturbance; the random combination of metrics can produce a final tool that better indicates disturbance. In the case of the spatial MMI, the complicated correlation of metrics and metric categories is controlled for in the selection process and the best model from a series of similar models is selected through statistical procedures.

One of the common criticism of landscape metrics is that while they are certainly useful in describing the spatial structure of patches within a landscape, there is no explicit connection between many of these metrics and any actual ecological processes (Voc et al., 2001). Since our intent is to use these metrics as indicators of disturbance in a multimetric index, the nebulous connection to ecological processes is unimportant, as long as the sum of metric scores is strongly and reliably indicative of disturbance. In this way, MMIs can function as useful tools in wetland management and conservation, even if we are uncertain of the causal mechanisms linking disturbance to the observed response in each metric (Schoolmaster et al., 2012). However, we can speculate about the reasoning behind the sensitivity of the metrics included in an MMI.

My 6-metric MMI included two metrics related to assemblages dominated by wetland-obligate *Carex* spp.   The percentage of the landscape (PLAND) and the splitting index (SPLIT) of vegetation assemblages dominated by wetland-obligate *Carex* spp. are reduced in wetlands experiencing greater agriculture-related disturbance. The percentage of the landscape metric is an area/edge metric that calculates the proportion of a landscape comprised of a particular class (McGarigal, 2015); in my case, the proportion of a wetland occupied by obligate *Carex* spp. The percentage of the landscape metric does not convey any information about the configuration or dispersion of *Carex* spp. within the wetland, only the percentage of the wetland that is occupied by wetland-obligate *Carex* species. In my sample sites, the percentage of the landscape metric was lower in high disturbance sites, with 12 of 18 sites in the highest quartile of disturbance (Disturbance score: 175-243) having no *Carex* dominated communities at all.   The mean percentage of the landscape of the sites that did have wetland obligate *Carex* communities was 14.4% with a median value of 8.8%. Comparatively, the wetlands in the lowest quartile in terms of their disturbance scores (Disturbance score: 2.8-96) had a mean percentage of the landscape metric value of 26.9% with a median percentage of the landscape metric value of 20.3%. This implies that low disturbance wetlands had a higher percentage of vegetation communities whose dominant vegetation type is a wetland-obligate *Carex* species.

The second metric associated with obligate *Carex* spp. was the splitting index, an aggregation metric which was originally created to assess habitat fragmentation measuring the probability of two animals in separate patches of the same vegetation assemblage meeting in the landscape (Jaeger, 2000). The splitting index essentially measures fragmentation by producing a the number of patches of equal size that a vegetation assemblage would need to be divided into to obtain the degree of fragmentation observed naturally (McGarigal, 2015). This metric ranges from 1 (where the landscape is a single patch) to the number of cells in the landscape squared (McGarigal, 2015). The splitting index value increases as the wetland obligate *Carex* spp. patches are reduced in area and sub-divided into smaller patches, therefore the metric score increases as the number of patches of *Carex* become less aggregated.

The metric associated with *Alisma triviale* dominated assemblage is the landscape shape index (LSI). The landscape shape index is a measure of the perimeter-area ratio for a given class within a landscape and the landscape boundary (McGarigal, 2015), which is the wetland edge plus the perimeter of *Alisma triviale* dominated patches divided by the landscape area. This metric was found to be positively associated with disturbance, meaning that as disturbance increased, the amount and complexity of *Alisma triviale* dominated patches also increased. Given that *Alisma triviale* is often found in mud flats (Stewart & Kantrud, 1972) and has been shown to be resistant to sedimentation caused by disturbance (Gleason et al., 2003), it stands to reason that *Alisma triviale* may be associated with agriculturally disturbed areas. As agricultural disturbance increases, so too does the complexity of *Alisma triviale* dominated communities.

The metric associated with *Salix* spp. dominated communities was the contrast-weighted edge density (CWED), which is negatively associated with disturbance. The contrast weighted edge density metric is a measure of edge density, i.e., the total edge of patches of a class divided by the landscape area (McGarigal, 2015). The contrast weighted edge density metric modifies this edge density value by using the edge contrast values from the edge contrast table (Appendix 3.4) taking into account the degree of similarity between the edge of the class and its neighbors (McGarigal, 2015). In my case, the edge contrast was based on differences between the growth form of neighboring communities. *Salix* spp. can grow to be several meters tall, giving them strong contrast with the forbs and grasses often found in wetland communities. As disturbance increases, the contrast weighted edge density value was observed to decrease meaning that the complexity and size of *Salix* spp. communities decreases with disturbance.

The final two metrics are calculated for the entire wetland, they are not associated with any specific vegetation assemblage. The first is the total edge (TE) within the landscape, which is sum of the length of every edge segment for every patch in the landscape (McGarigal, 2015). This metric will increase in value as more edge is present in the landscape (McGarigal, 2015), therefore the greater the number of patches and the more complex the shape of those patches, the higher the total edge value will be. The total edge metric is negatively associated with disturbance, meaning that low disturbance wetlands have a greater number of patches and greater patch shape complexity than high disturbance wetlands. High disturbance wetlands have fewer patches with less complex edges.

The second landscape metric is the aggregation index (AI), an aggregation metric that calculates the number of like adjacencies (cells entirely surrounded by cells of the same vegetation assemblage) for each vegetation assemblage as a proportion of the total possible number of like-adjacencies (if all the patches were aggregated into one patch) summed for each vegetation assemblage in the landscape (McGarigal, 2015). This metric represents the aggregation of patches on the landscape as a percentage value; the AI would be 0 if there were no like adjacencies and 100 if the landscape was entirely one patch (McGarigal, 2015). This metric is positively associated with disturbance, meaning that as disturbance increases, the vegetation assemblages within a wetland become increasingly aggregated.

Based on the trends observed in the spatial metrics, low disturbance wetlands are characterized by a high number of obligate *Carex* patches (PLAND) that are spread out through the wetland (SPLIT). Also, complex patches of *Salix* shrubs and simple to no patches of *Alisma triviale* (LSI). Finally, low disturbance wetlands had a greater variety of other vegetation assemblage patches (TE) that are dispersed with low aggregation (AI). These characteristics were found in low disturbance wetlands using the 6-metric MMI based on spatial metrics.

An effective MMI must apply equally well across its jurisdiction of implementation, must provide consistent scores that are insensitive to interannual variation in climate, and must be unbiased in terms of the scores achievable by marshes of differing hydroperiod or permanence class. The results of the ANOVA and subsequent Tukey's test indicate that there is a marginally significant difference between wetland classes, specifically temporary wetlands. This difference is likely due to the difference in size of temporary wetlands compared to larger wetlands with longer hydroperiods. Smaller wetlands will have less complex zonation, small assemblages and lower assemblage richness based on field observations. While this tool remains effective in temporary wetlands, it would be somewhat misleading to compare MMI scores between temporary wetlands and wetlands of a longer hydroperiod. The non-significant results for the two-sample t-tests show that the spatial MMI is robust to inter-annual variation and differences in ecoregion.

One metric, the total edge of vegetation assemblages in the wetland, was found to have a strong relationship with area. The values for total area would increase as wetland size increased since total edge not only measures the edge of vegetation patches within the wetland but of the edge of the wetland itself (McGarigal, 2015). This trend is likely responsible for the strong area association seen between temporary wetlands and the spatial MMI scores as temporary wetlands tend to be smaller than wetlands of a longer permanence class. Though this does not significantly compromise the use of the MMI in temporarily wetlands, the implication is that care should be exercised in comparing temporarily-ponded wetland MMI scores to the scores of more permanently-ponded wetlands and that small wetlands may be undervalued. This is a risk because research has shown that small wetlands are already experiencing greater rates of loss than larger wetlands in Parkland Alberta (Serran & Creed, 2014). This association with wetland area brings a caveat to the use of the spatial MMI meaning that the 6-metric MMI will likely produce a lower score for temporary wetlands and should be substituted with the 4-metric spatial MMI when looking at temporary wetlands.

3.5 Conclusion

To address the problem of continuing wetland loss, the Government of Alberta has instituted a new wetland policy aimed at mitigating wetland loss while providing for removal of wetlands should there be no alternative. Under this policy, wetland removal requires the replacement of lost ecosystem services, which often requires the creation or restoration of a wetland. To evaluate the success or failure of wetland restoration a scientifically validated tool is needed to assess wetland health. This chapter details the creation and validation of a multimetric index that measures the health of wetlands using the novel approach of having the spatial characteristics of wetland vegetation communities be the biotic indicator of wetland health.

Using spatial metrics to act as an indicator of disturbance is a new approach within the Prairie Pothole wetland ecosystem. My results show that it is possible to make an MMI using landscape ecology metrics on the individual wetland scale. This type of assessment has the potential to be carried out using remote sensing or UAV's which would significantly increase the efficiency with which these assessments could be carried out, which in turn could lead to an increased monitoring and assessment put forth by managers and stakeholders. In addition, the metrics used in this tool could provide insight to the restoration agents in designing newly restored wetlands. Knowing that low-disturbance wetlands tend to have less aggregated, more complex sedge communities could lead to better planning and planting of wetland vegetation communities to attempt to mimic those seen in low-disturbance wetlands.

3.6 Figures

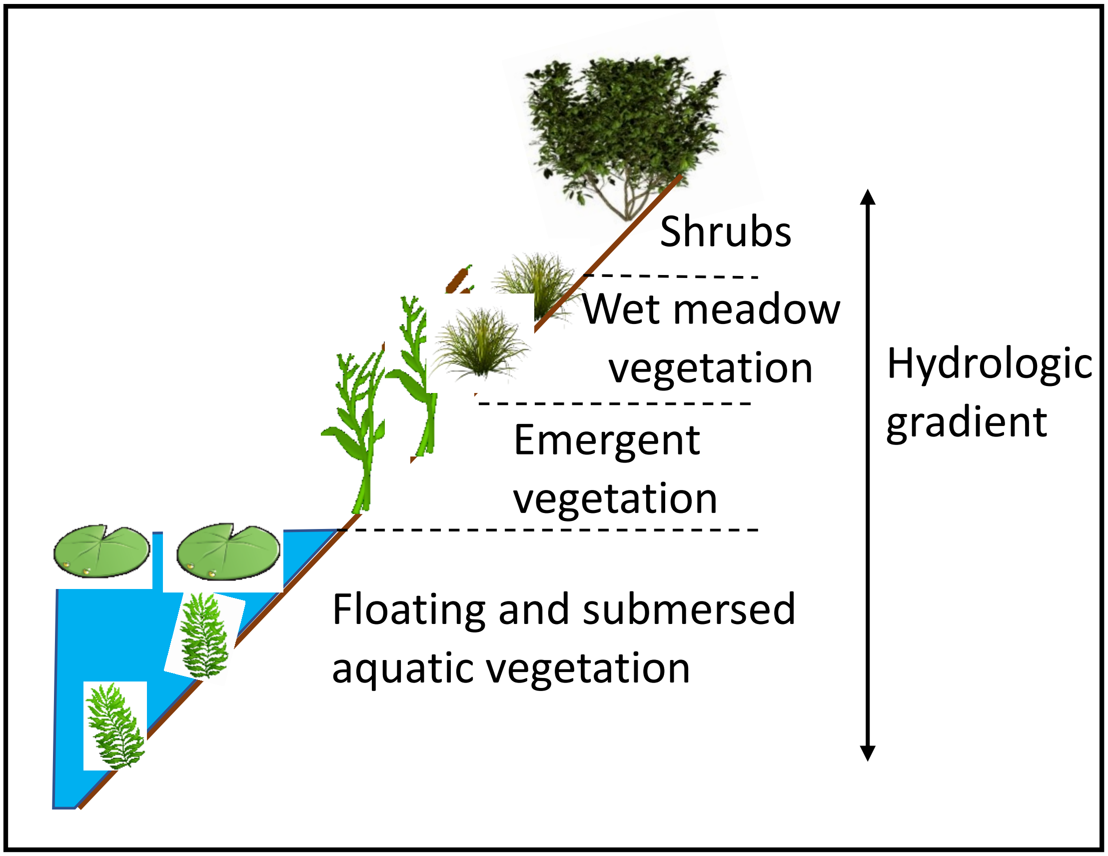

Figure 3-1: A schematic of different wetland vegetation assemblages along a hydrologic gradient. Adapted from Keddy (2000). Note that the slope of the shoreline is not to scale.

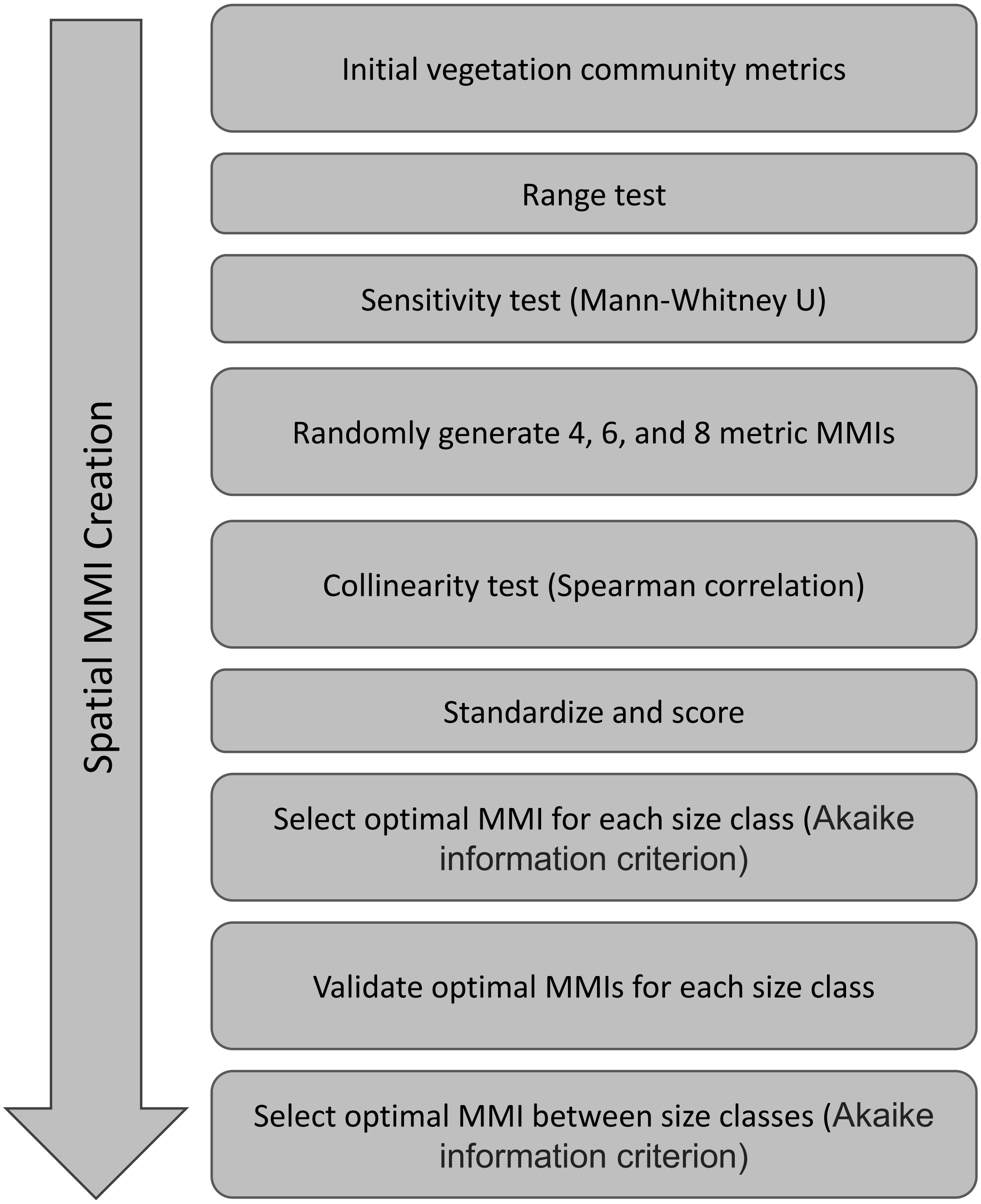

Figure 3-2: A schematic summarizing the steps in creating my spatial MMI and the statistical procedures used at each step.

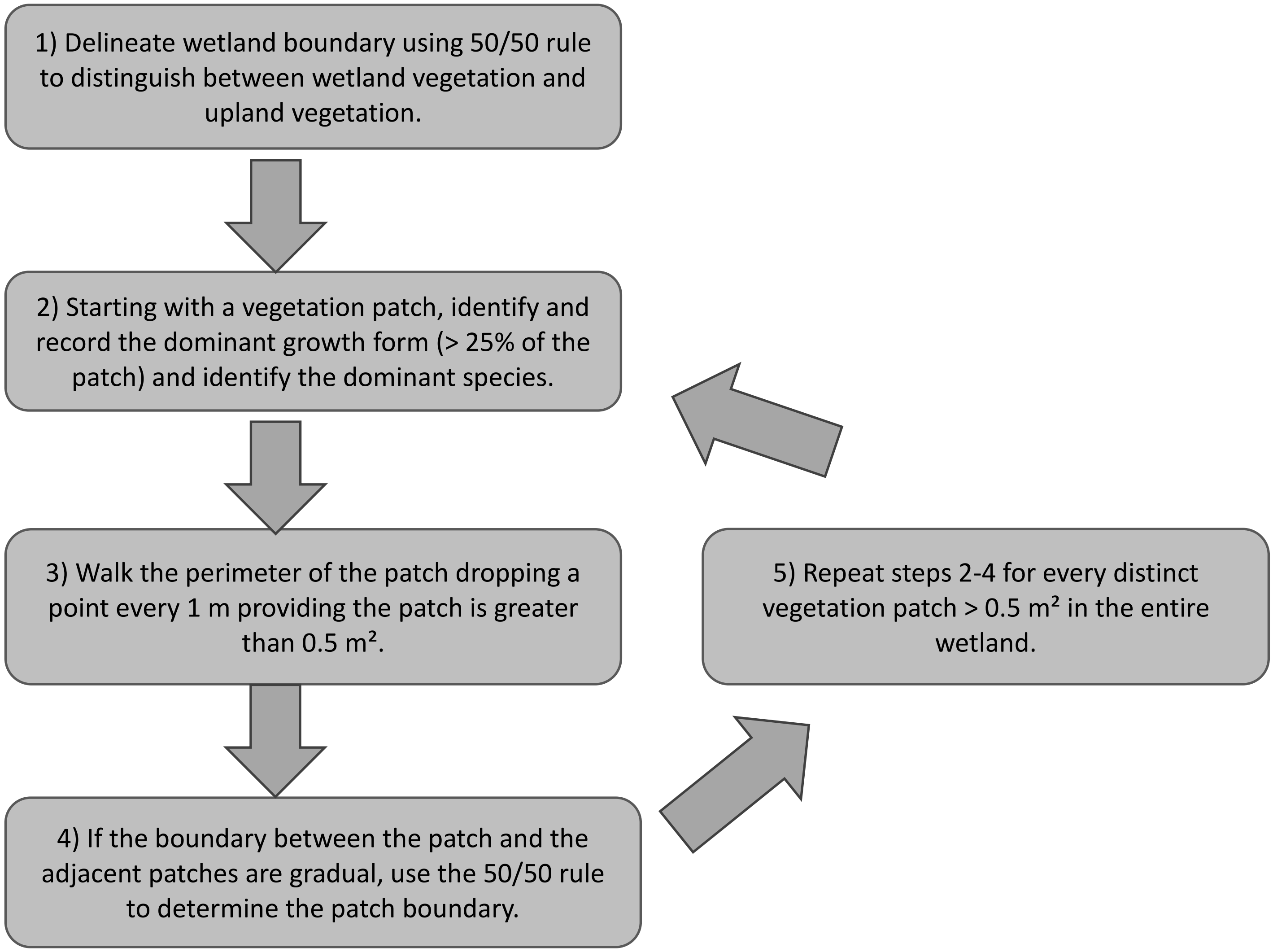

Figure 3-3: Schematic outlining the steps in delineating and mapping wetland vegetation communities using the 50/50 rule discussed in the methods section.

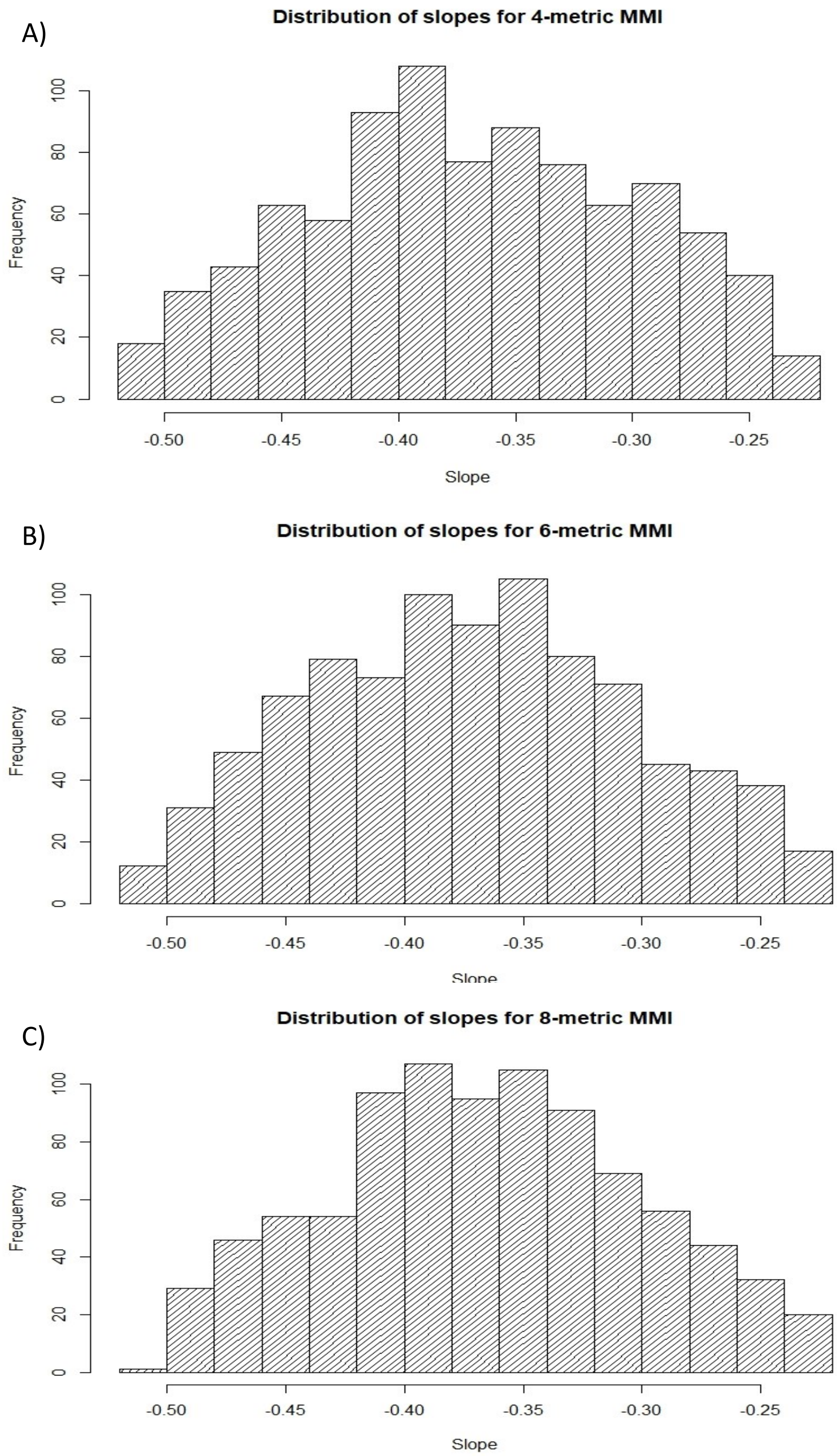

Figure 3-4: Histograms showing the distribution of slope values calculated through bootstrapping without replacement (k=1000). Distributions for the optimal 4-metric MMI (A), the optimal 6-metric MMI (B), and the only acceptable 8-metric MMI (C) are all less than zero.

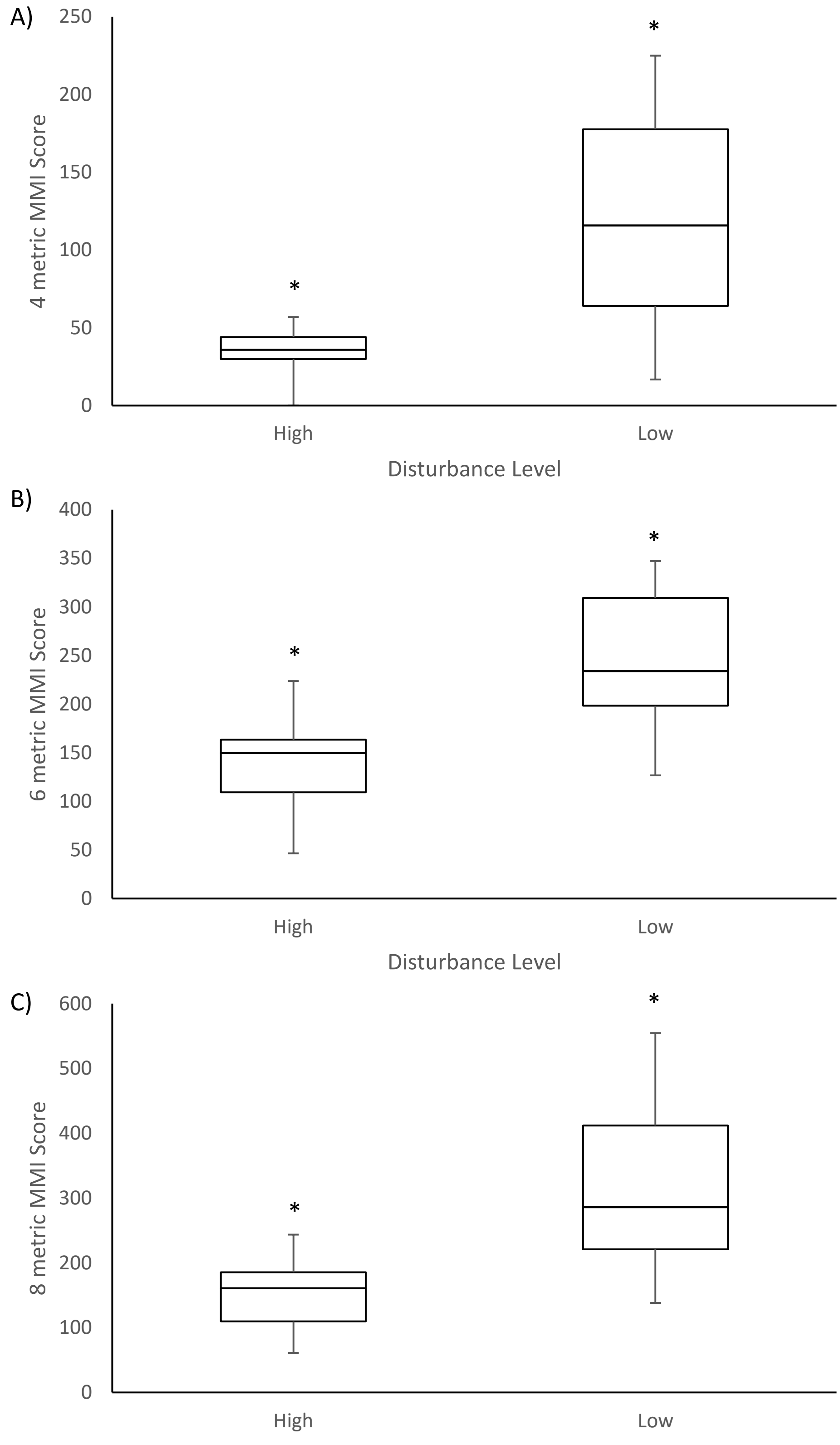

Figure 3-5: Box plots for MMI scores at low (n = 13) and high (n = 13) disturbance sites for A) 4-metric, B) 6-metric and C) 8-metric MMI with asterisk (*) indicating a significant difference between high and low disturbance according to Mann- Whitney U test.

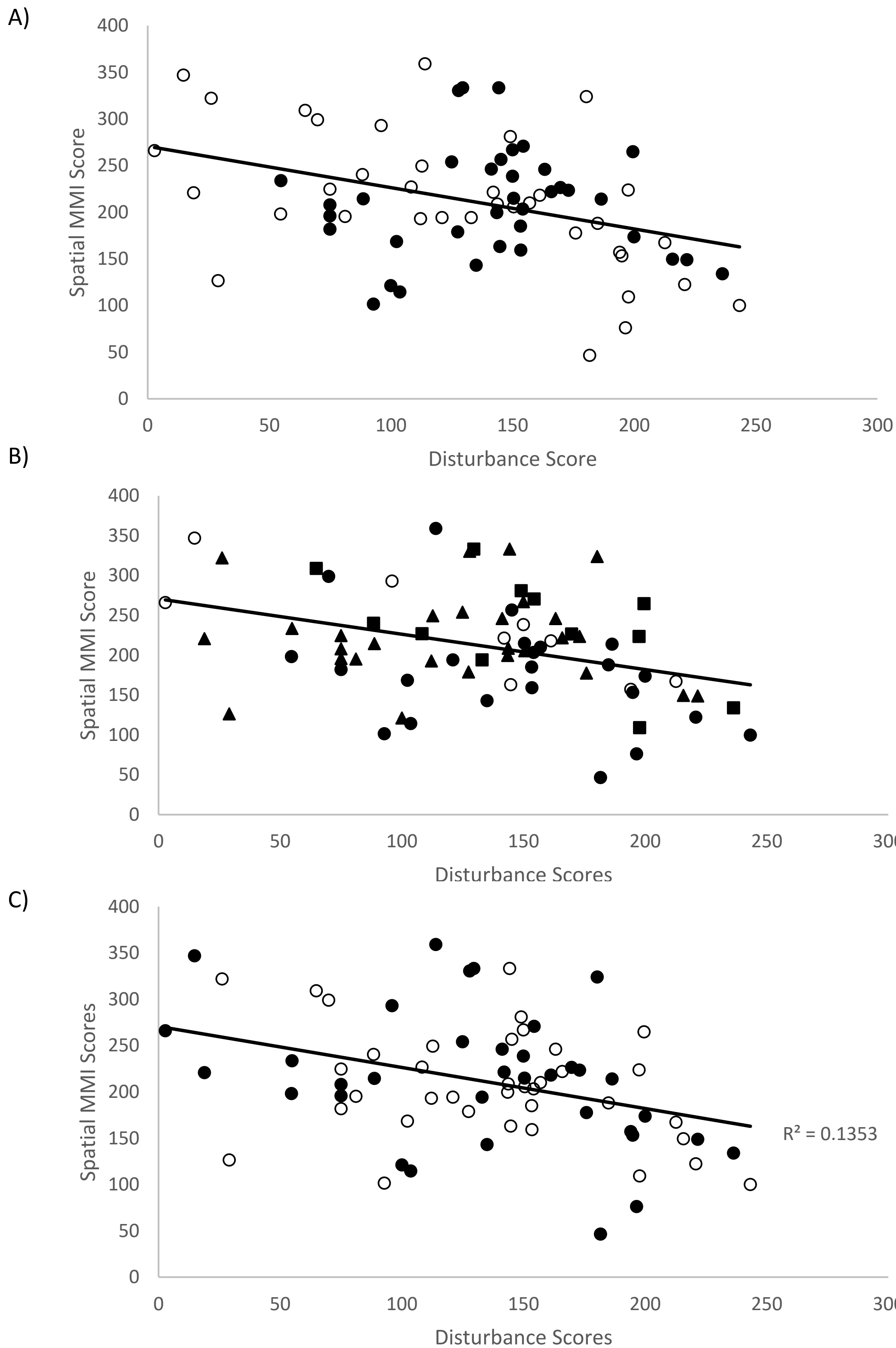

Figure 3-6: Scatter plot of final spatial MMI scores and disturbance scores coded by: A) region with the grassland as solid circles and the parkland as open circles, B) wetland permanence class with solid circles as temporary, triangles as seasonal, squares as semi-permanent and open circles as permanent wetlands, and C) sample year with 2014 sites as solid circles and 2015 sites as open circles.

3.7 Tables

Table 3-1: A summary of different spatial metric groups, a brief description and the level at which the metric is applicable: patch (referring to an individual patch of vegetation), vegetation assemblage (all the patches of the same type considered together) or the entire wetland level. Adapted from (McGarigal & Marks, 1994).

Metric Group	**Description**	**Applicable Level**
Area/Edge	Metrics that describe the size of patches through their area and the amount of edge around the patches.	Patch, vegetation assemblage & wetland
Shape	Estimates of patch shape based on various calculations using the area and perimeter of patches.	Patch, vegetation assemblage & wetland
Core Area	Metrics based on the difference between edge habitat and habitat at the center, or core, of a patch.	Patch, vegetation assemblage & wetland
Contrast	Metrics related to the difference in habitat between all possible pair-wise combinations of vegetation assemblages in the wetland.	Patch, vegetation assemblage & wetland
Aggregation	Metrics that describe the spatial arrangement of patches across the landscape.	Vegetation assemblage & wetland
Diversity	Metrics that quantify the differences in patch composition through standard richness measures and diversity indices such as Shannon's and Simpson's diversity indices.	Wetland

Table 3-2: The top ten MMI models in the 4- and 6-metric category based on their AICc values and AICc weights and the only 8-metric model.

Number of Metrics	Metrics	AICc	Δ AICc	Model Likelihood	AICc Weights	Cumulative AICc Weights
4-Metrics	oblcarex_TE, oblcarex_SPLIT, salix_ED, AI_Land	774.66	0	1	0.12	0.12
	oblcarex_ED, alisma_TE, salix_ED, NDCA_Land	774.72	0.06	0.97	0.12	0.24
	oblcarex_TE, oblcarex_SPLIT, salix_ED, AI_Land	775.03	0.37	0.83	0.1	0.35
	oblcarex_PD, alisma_COHESION, salix_ED, NDCA_Land	775.03	0.37	0.83	0.1	0.45
	oblcarex_ED, alisma_MESH, salix_LPI, NDCA_Land	775.04	0.39	0.82	0.1	0.55
	AI_Land, alisma_PLAND, salix_ED, TE_Land	775.13	0.47	0.79	0.1	0.65
	oblcarex_NDCA, alisma_PLADJ, salix_ED, AI_Land	775.32	0.66	0.72	0.09	0.74
	ED_Land, alisma_ED, salix_PLAND, TE_Land	775.38	0.72	0.7	0.09	0.83
	TE_Land, alisma_LPI, salix_LPI, ED_Land	775.38	0.72	0.7	0.09	0.91
	oblcarex_ED, alisma_LPI, salix_ED, LSI_Land	775.39	0.73	0.69	0.09	1

Number of Metrics	Metrics	AICc	Δ AICc	Model Likelihood	AICc Weights	Cumulative AICc Weights
6-Metrics	oblcarex_PLAND, oblcarex_SPLIT, alisma_LSI, salix_CWED, TE_Land, AI_Land	773.24	0	1	0.19	0.19
	oblcarex_PLAND, oblcarex_SPLIT, alisma_LPI, salix_PLAND, LSI_Land, AI_Land	773.85	0.61	0.74	0.14	0.33
	oblcarex_TE, oblcarex_SPLIT, alisma_NLSI, salix_ED, ED_Land, AI_Land	774.39	1.15	0.56	0.11	0.44
	oblcarex_PLAND, oblcarex_SPLIT, alisma_SPLIT, salix_DCAD, NDCA_Land, AI_Land	774.55	1.32	0.52	0.1	0.53
	oblcarex_PLAND, oblcarex_PD, alisma_NDCA, salix_ED, TE_Land, AI_Land	774.59	1.35	0.51	0.1	0.63
	oblcarex_PLAND, oblcarex_NDCA, alisma_PLAND, salix_ED, TE_Land, AI_Land	775	1.76	0.41	0.08	0.71
	oblcarex_PLAND, oblcarex_LPI, alisma_SPLIT, NDCA_Land, ED_Land, AI_Land	775.05	1.81	0.4	0.08	0.78
	oblcarex_LPI, oblcarex_NP, alisma_LPI, salix_CWED, NDCA_Land, AI_Land	775.08	1.84	0.4	0.08	0.86
	oblcarex_PD, oblcarex_AI, alisma_NP, salix_ED, TE_Land, AI_Land	775.17	1.93	0.38	0.07	0.93
	oblcarex_ED, oblcarex_NP, alisma_ED, salix_PLAND, LSI_Land, AI_Land	775.3	2.06	0.36	0.07	1
8-Metrics	alisma_DIVISION, oblcarex_TECI, oblcarex_DCAD, salix_LSI, oblcarex_SPLIT, LSI_Land, AI_Land, oblcarex_MESH	NA	NA	NA	NA	NA

Table 3-3: Spatial MMI metrics and their properties for the optimal MMI of each size class (4, 6, & 8). Detailed descriptions of metrics can be found in Appendix 3.1.

MMI	Metric Names	Descriptions	Relationship to Disturbance	Spatial Metric Group	Spearman Rho	p-value
4-Metric MMI	ne.oblcarex_TE	Total edge length of obligate *Carex* communities	negative	Area/Edge	-0.0218	0.8832
	ne.oblcarex_SPLIT	The splitting index for obligate *Carex* communities	negative	Aggregation	-0.2009	0.0906
	AI_Land	The aggregation index for all vegetation assemblages across the entire landscape	positive	Aggregation	0.1866	0.1165
	ts.salix_ED	Total edge density for *Salix* communities	negative	Area/Edge	-0.1558	0.1912
6-Mertic MMI	ne.oblcarex_PLAND	Percentage of the landscape occupied by obligate *Carex* communities	negative	Area/Edge	-0.1869	0.1159
	be.alisma_LSI	Landscape shape index for *Alisma triviale* communities	positive	Aggregation	0.1761	0.1389
	ts.salix_CWED	Contrast-weighted edge density of *Salix* communities	negative	Contrast	-0.1510	0.2056
	TE_Land	Total edge of all vegetation assemblages across the landscape	negative	Area/Edge	-0.0926	0.4393
	ne.oblcarex_SPLIT	The splitting index for obligate *Carex* communities	negative	Aggregation	-0.2009	0.0906

MMI	Metric Names	Descriptions	Relationship to Disturbance	Spatial Metric Group	Spearman Rho	p-value
6-Metric MMI	AI_Land	The aggregation index for all vegetation assemblages across the entire landscape	positive	Aggregation	0.1866	0.1165
8-Metric MMI	be.alisma_DIVISION	Landscape division index for *Alisma triviale* communities	positive	Aggregation	0.1791	0.1322
	ne.oblcarex_TECI	Total edge contrast index for obligate *Carex* communities	positive	Contrast	0.2714	0.0644
	ne.oblcarex_DCAD	The disjunct core area density of obligate *Carex* communities	negative	Core Area	-0.2859	0.0149
	ts.salix_LSI	Landscape shape index for *Salix* communities	negative	Aggregation	-0.1367	0.2522
	ne.oblcarex_SPLIT	The splitting index for obligate *Carex* communities	negative	Aggregation	-0.2009	0.0906
	LSI_Land	The landscape shape index for all vegetation assemblages across the entire landscape	negative	Aggregation	-0.1824	0.1251
	AI_Land	The aggregation index for all vegetation assemblages across the entire landscape	positive	Aggregation	0.1866	0.1165
	ne.oblcarex_MESH	The effective mesh size for obligate *Carex* communities	negative	Aggregation	-0.1629	0.1717

Table 3-4: 90% confidence intervals for 4, 6, and 8-metric MMI bootstrapped slope values.

MMI	Lower 90% CI	Upper 90% CI	Mean Slope
4-Metric	-0.5079	-0.2278	-0.36842
6-Metric	-0.5121	-0.2291	-0.36963
8-Metric	-0.5009	-0.2311	-0.36637

Table 3-5: Results for Mann-Whitney U test for validation of optimal MMIs of 4, 6, and 8 metrics by testing for a significant difference in MMI scores between the sites from the upper and lower quintile of disturbance.

MMI	Mann-Whitney U Statistic	p-value	Median for high disturbance	Median for low disturbance	Degrees of freedom for high disturbance	Degrees of freedom for low disturbance
4-metric	21	< 0.001	35.8062	114.7821	13	13
6-metric	12	< 0.001	149.6131	233.8040	13	13
8-metric	20	< 0.001	160.8817	286.0755	13	13

Table 3-6: Results for final comparison between optimal spatial MMIs within each metric class performed using AICc.

MMI size class	AICc	Δ AICc	Model Likelihood	AICc Weights	Cumulative AICc Weights
6-metric MMI	773.24	0	1	0.66	0.66
4-metric MMI	774.66	1.42	0.49	0.32	0.98
8-metric MMI	780.03	6.8	0.03	0.02	1

Table 3-7: Results of analysis of variance to detect any significant difference among spatial MMI scores by wetland permanence classes.

Source	Type III sum of squares	Degrees of freedom	Mean Squares	F-Ratio	p-Value
Permanence Class	40,468.82	3	13,489.61	3.335	0.024
Error	275,078.89	68	4,045.28		

Table 3-8: Results of Tukey's honestly-significant-difference test comparing MMI scores of wetlands by different permanence classes.

Permanence class (i)	Permanence class (j)	Difference	p-Value	95% Confidence Interval	
				Lower	Upper
Temporary	Seasonal	-45.458	0.063	-92.598	1.682
Temporary	Semi-permanent	-57.786	0.061	-117.438	1.866
Temporary	Permanent	-53.481	0.151	-119.344	12.381
Seasonal	Semi-permanent	-12.328	0.943	-70.125	45.469
Seasonal	Permanent	-8.024	0.988	-72.211	56.164
Semi-permanent	Permanent	4.304	0.999	-69.562	78.171

Table 3-9: Results of a two-sample t-test for a difference in MMI scores between natural regions or between sampling years.

	t-statistic	**Degrees of freedom**	**p-value**
Natural region	-0.226	70	0.822
Sampling year	0.415	70	0.680

4. Conclusion

4.1 Overview

Wetlands in the prairie pothole region of North America provide a number of important ecosystem services (Bartzen et al., 2010), including but not limited to supporting biodiversity, regulating weather events through flood mitigation and ground water sequestration, and the filtration and removal of pollutants from water (Bartzen et al., 2010; Beyersbergen et al., 2004). Despite these benefits, prairie pothole wetlands are often removed from a landscape in favor of expanding agriculture or industry (Davidson, 2014), and it is only recently that the impact of the loss of these wetlands has become a management priority.

In the province of Alberta, legislators have moved forward with policy to protect and restore wetlands with the aim of retaining the ecosystem services they provide (Government of Alberta, 2013). Integral to this policy is the need for assessment and evaluation of wetlands, which includes wetlands that will be directly impacted by development and those being created to mitigate wetland loss (Government of Alberta, 2013, 2016). One of most common tools used in these ecological assessments is the multimetric index (MMI) (Barbour & Yoder, 2000), which uses the response of a biotic community to disturbance as an indicator of wetland condition. My work addresses the need for monitoring and assessment tools in Alberta and explores the methods for building a multimetric assessment. The goals of my thesis were to 1) construct a floristic-composition based MMI comparing the traditional method of MMI development with an iterative method proposed by van Sickle (2010), and 2) to construct an MMI using the response of the spatial arrangement of wetland vegetation communities as an indicator of disturbance.

4.2 Research Findings

In Chapter 2 of my thesis, I developed an MMI based on the floristic composition of marshes in the Parkland and Grassland ecoregions of Alberta, the jurisdiction of the province that is managed as the "white zone." In the process of constructing this tool, I compared two different methods for building an MMI: the traditional method of using correlation to guide metric selection, and an emerging method using random selection to generate a number of potential MMIs and selecting the optimal MMI from that pool. Using both methods I was able to develop and successfully validate an MMI based on measurements of floristic community composition, and determined that the iterative method produced a tool that was more strongly indicative of agriculture-related disturbance than the traditional method.

In Chapter 3 of my thesis, I developed an MMI using the spatial arrangement of vegetation assemblages to indicate disturbance in the "white zone." The composition and arrangement of wetland vegetation is affected by disturbance both within and outside the wetland, thus I hypothesized that the arrangement of wetland vegetation communities could be used as an indicator in an MMI. Using the iterative method described in Chapter 2, I successfully developed and validated an MMI using spatial metrics for vegetation communities.

4.3 Implications and Significance

The successful development of two vegetation-based MMIs will facilitate wetland assessment and monitoring in the Prairie and Parkland regions of Alberta, and directly contributes to the goals of the Alberta wetland policy. While vegetation-based MMIs have been developed in Alberta previously, they were limited to specific ecoregions or wetlands of a specific hydroperiod. The MMIs developed in my thesis provide a means to assess the condition of marshes in the white zone with a single tool, as both the MMIs based on floristic composition and the spatial arrangement of vegetation patches were developed to be used in the Parkland and Grassland region across a gradient of wetland hydroperiod from temporarily-ponded to permanently-ponded marshes. Importantly, I found that the iterative method of developing an MMI produced a tool that better indicated disturbance when compared with the traditional method. Additional benefits of using the iterative method include that the random metric-selection process removes any subjective biases in metric selection, and that the code used to conduct metric selection makes the process repeatable. This provides an adaptable method for future MMI construction, and builds on the methods proposed by van Sickle (2010).

The floristic composition based MMI is a broadly applicable, easy to use tool to assess wetland condition. Since all of the metrics are presence/absence measures, any site assessments only require richness measures rather than accurate estimates of relative abundance. A potential drawback of this tool includes the time it takes to perform a site assessment, which can vary depending on the complexity and species richness of the vegetation community. To perform the assessment in the field, adequate botanical knowledge is required to delineate vegetation community boundaries and identify species present in sampling quadrats. This necessity presents a moderate barrier, as sampling must be restricted to periods of time during which vegetation is identifiable (i.e., when flowers or fruit are evident).

The MMI based on the spatial arrangement of vegetation patches is applicable across the same region and wetland types as the floristic composition based MMI, and also requires the delineation of wetland vegetation patch boundaries. However, the spatial metrics do not require such detailed botanical knowledge as the metrics based on floristic composition. However, this approach requires a high-precision GPS unit and GIS software to process the field data. Attempting to implement this tool remotely based on high resolution imagery would also require the development of image classification techniques capable of accurately delineating vegetation patches. This presents an opportunity for rapid, reliable assessment of wetlands and reducing the need for intensive field surveys. Ultimately the choice between either MMI depends on the resources available to the practitioner and the requisite accuracy of the assessment. Whereas the MMI based on vegetation community composition is more labor intensive, it is also a slightly better indicator of agriculture-related disturbance.

The method of validation differed between the floristic and spatial MMI with the floristic MMI validated using a leave-p-out method, and the spatial MMI validated using a combination of bootstrapping without replacement and a Mann-Whitney U test. This was done because the spatial metrics were shown to have a lower signal to noise ratio than the floristic metrics (Table 4-1). This highlights an important distinction between the spatial and the floristic MMIs. The floristic MMI has a stronger R-squared when compared to the disturbance scores than the spatial MMI (Figure 4-1). The higher signal to noise ratio for the floristic MMI shows that the floristic MMI has a higher signal than the spatial MMI which has more noise obscuring the signal. Both tools were validated and suitable for use, however the floristic MMI better indicated disturbance than the spatial MMI.

The disturbance scores used in the creation of both MMIs were derived from land-cover data and modified using assigned values based on field observations. This method of disturbance score calculations is less quantitative than other methods of creating disturbance scores in this region (e.g., Raab & Bayley, 2012; Rooney & Bayley, 2011; Wilson, Bayley, et al., 2013). The land cover data lacks the resolution to properly identify grazing habitat and the intensity of land use. As well, the modifications made to the extent of non-natural land cover to incorporate field observations of grazing, the lack of riparian buffers, or the presence of pesticides are non-continuous and somewhat arbitrary. Despite this, I was able to successfully validate multiple tools using this disturbance gradient and there was no discernable difference in disturbance scores between regions or sampling year (Table 2-7). Given the nature of the disturbance scores, it is possible that the error variance in regressions between MMI and disturbance scores (i.e., the relative low R squared values for both the floristic and spatial MMIs) might reflect inaccuracy in the disturbance scores rather than in the MMI scores. Since the MMIs are created using the response of vegetation within the sample wetlands, it is possible that the MMIs are better indicating the true level of disturbance affecting a wetland than the coarse measure reflected in the disturbance scores.

4.4 Future Work

The successful use of the spatial arrangement of vegetation patches as an indicator of disturbance provides a faster, less fieldwork-intensive method of developing an MMI. It also lays the groundwork for future work in this area that would incorporate remote sensing technology to accurately distinguish and delineate wetland vegetation communities and identify the dominant species present in these communities. The ability to use remote sensing to measure wetland integrity presents the possibility of rapid, remote assessments that are scientifically valid and reliable. Some challenges exist before remote sensing can be successfully incorporated into the development of a spatial MMI. Mainly, the data collected through remote sensing must undergo ground-truthing to establish if remote sensing can accurately delineate wetland vegetation community boundaries, identify the growth form and dominant species in a vegetation community, and that data aligns with field-mapped vegetation communities. The spatial MMI has two metrics that are based on wetland obligate *Carex* spp. communities. It is important that any remote method be able to distinguish these wetland obligate species from other *Carex* species that could occur in NPPR marshes. These issues would need to be addressed before a UAV or other remote sensing method could be relied on to calculate these MMI scores.

While there is the potential to develop a rapid assessment tool using remote sensing, this method could not totally replace field-based site assessments. I recommend that the floristic MMI is used to carry out wetland assessments for the purposes of addressing the needs presented by the Alberta wetland policy and wetland restoration directive (Government of Alberta, 2013, 2016). This approach explains more variance in disturbance scores, has a greater signal to noise ratio, and provides a more fine-scale measurement of the vegetation community (Figure 4-1). If the resources are available to develop the MMI based on the spatial arrangement of vegetation patches, I would recommend that both MMIs be used as complementary tools. Given the stronger signal to noise ratio of the floristic MMI, it is the better option for an assessment tool, though it requires a stronger taxonomic knowledge and the sampling period is limited to the flowering period of most wetland vegetation. By comparison, the spatial MMI, though it has lower signal, could be carried out in a wider timeframe and requires less taxonomic knowledge than the floristic MMI. With this in mind, the more intensive MMI based on floristic composition could be used to evaluate the baseline and closure condition of a wetland and the more synoptic MMI based on the spatial arrangement of vegetation patches could be used in the intervening years as a monitoring tool to gauge the progress of wetland restoration, analogous to the USEPA level 2 and level 3 assessments (U.S.EPA, 2017).

Both the floristic and spatial MMIs provide further insight into wetland restoration. While sedges are associated with wetlands in this region (Stewart & Kantrud, 1971), the spatial MMI indicates that low disturbance wetlands have larger patches of wetland-obligate *Carex* spp., while higher disturbance wetlands have smaller, more dispersed patches of wetland-obligate *Carex* spp. This reveals that successfully restored wetlands should have larger, more aggregated patches of wetland-obligate sedges. As well, the presence of a metric associated with the arrangement of *Salix* shrubs suggests that restoration would be aided by the planting of willows in restored wetlands, such that they yielded meandering patches of high spatial complexity. Thus, metrics from both MMIs can be used to derive guidance for wetland restoration.

Previous work in this region used avian communities as an indicator of wetland disturbance (Anderson, 2017) and an MMI was developed and validated for the Parkland region using metrics based on the avian community. There likely exists a relationship between the spatial arrangement of wetland vegetation communities and avifauna. For example waterfowl will select wetlands based on the ratio of emergent vegetation to open water (Poiani & Johnson, 2012). The results of the floristic and spatial MMIs can be used to elucidate the relationship between avian and vegetation communities, both in terms of the floristic composition and the spatial arrangement of wetland vegetation communities. An improved understanding of the relationship between birds and vegetation could also provide guidance to restoration agents.

As human expansion and development is likely to continue in Alberta, efforts must be taken to preserve the ecosystem services provided by Alberta wetlands. My work provides reliable, easy to use tools that can be used to assess wetlands that have been created to replace lost ecosystem services.

4.5 Figures

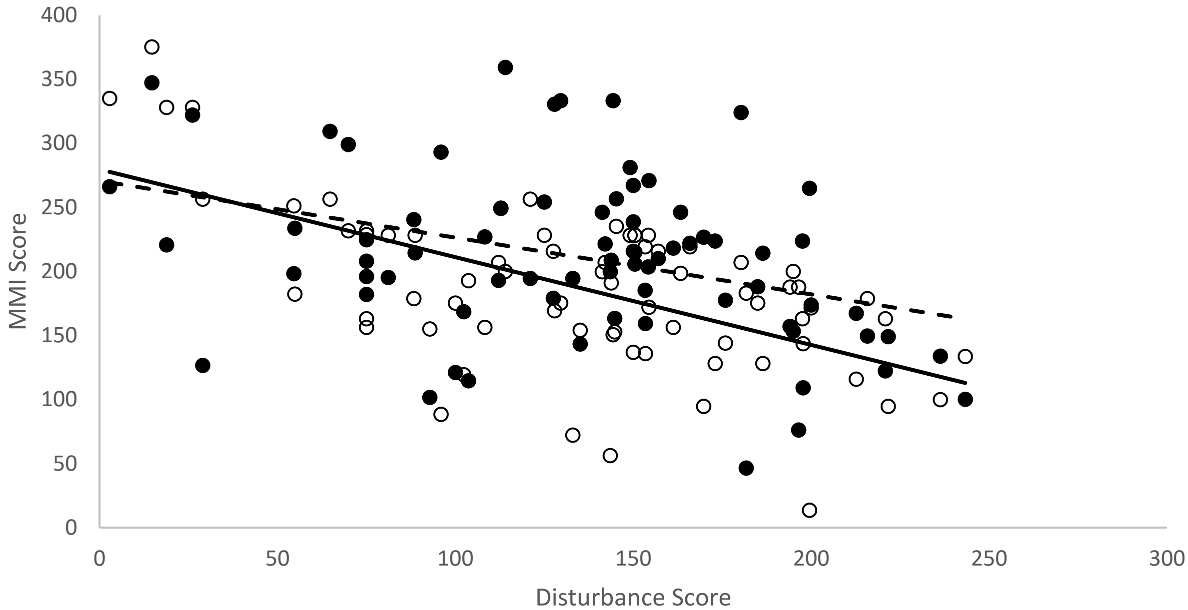

Figure 4-1: Linear regression of MMI scores with disturbance scores. The MMI based on floristic metrics had an $R^2 = 0.3655$ when regressed with disturbance (open circles, solid line), and the MMI based on spatial metrics had an $R^2 = 0.1353$ when regressed with disturbance scores (solid circles, dashed line).

4.6 Tables

Table 4-1: The signal to noise ratio for the spatial and floristic calculated by taking the ratio of the predicted mean over the standard deviation of the residuals for both tools.

MMI	Signal/Noise Ratio
Spatial	1.48
Floristic	3.02

References

AAFC. (2013a). Annual Crop Inventory 2011. Earth Observation Team, Science and Technology Branch, Agriculture and Agri-Food Canada. Retrieved from http://open.canada.ca/data/en/dataset/58ca7629-4f6d-465a-88eb-ad7fd3a847e3

AAFC. (2013b). Annual Crop Inventory 2012. Earth Observation Team, Science and Technology Branch, Agriculture and Agri-Food Canada. Retrieved from http://open.canada.ca/data/en/dataset/621bb298-116f-4931-8350-741855b007bc

AAFC. (2013c). Annual Crop Inventory 2013. Earth Observation Team, Science and Technology Branch, Agriculture and Agri-Food Canada. Retrieved from http://open.canada.ca/data/en/dataset/4b1d45b0-5bfe-4c6d-bcd3-96c9d821ad3b

AAFC. (2015). Annual Crop Inventory 2014. Earth Observation Team, Science and Technology Branch, Agriculture and Agri-Food Canada. Retrieved from http://open.canada.ca/data/en/dataset/ae61f47e-8bcb-47c1-b438-8081601fa8fe

AEP. (2014). Alberta Merged Wetland Inventory. Retrieved from https://geodiscover.alberta.ca/geoportal/catalog/search/resource/details.page?uuid=%257BA73F5AE1-4677-4731-B3F6-700743A96C97%257

Alberta Biodiversity Monitoring Institute (ABMI). (2017). Marsh Skullcap (Scutellaria galericulata). Retrieved from abmi.ca/home/dataanalytics/biobrowser-home/species-profile?tsn=99002674

Alberta Enviornment and Sustainable Research Development (ESRD). (2015). *Alberta Wetland Classification System*. Edmonton, AB.

Anderson, D. (2017). *Monitoring Wetland Integrity and Restoration Success with Avifauna in the Prairie Pothole Region of Alberta, Canada*. University of Waterloo.

Andreas, B. K., Mack, J. J., & Mccormac, J. S. (2004). Floristic quality assessment index (FQAI) for vascular plants and mosses for the State of Ohio.

Argus, G. W. (2008). A guide to the identification of Salix (willows) in Alberta. *Prairie and Northern Plant Diversity Centre, Workshop on Willow Identification*. Jasper National Park, Alberta.

Attrill, M. ., & Rundle, S. . (2002). Ecotone or ecocline: ecological boundaries in estuaries. *Estuarine, Coastal and Shelf Science*, *55*(6), 929–936. http://doi.org/10.1006/ecss.2002.1036

AWRAC. (2014). *Alberta Invasive Plant Identification Guide*.

Bailey, R. C., Linke, S., & Yates, A. G. (2014). Bioassessment of freshwater ecosystems using the Reference Condition Approach: comparing established and new methods with common data sets. *Freshwater Science*, *33*(4). http://doi.org/10.1086/678771.

Barbour, M. T., Stribling, J. ., & Karr, J. R. (1995). Multimetric Approach for Establishing Biocriteria and Measuring Biological Condition. In W. S. Davis & T. P. Simon (Eds.), *Biological Assessment and Criteria Tools for Water Resource Planning and Decision Making* (pp. 63–81). Lewis Publishers.

Barbour, M. T., & Yoder, C. O. (2000). The multimetric approach to bioassessment, as used in the United States of America. In J. F. Wright, D. W. Sutcliffe, & M. T. Furse (Eds.), *Assessing the biological quality of fresh waters: RIVPACS and other techniques.* (pp. 281–292). Oxford, UK.

Bartzen, B. A., Dufour, K. W., & Clark, R. G. (2010). Trends in agricultural impact and recovery of wetlands in prairie Canada, *20*(2).

Bell, S. S., Fonseca, M. S., & Motten, L. B. (1997). Linking Restoration and Landscape Ecology. *Restoration Ecology*, *5*(4), 318–323. http://doi.org/10.1046/j.1526-100X.1997.00545.x

Bettinger, P., Bradshaw, G. A., & Weaver, G. W. (1996). Effects of geographic information system vector-raster-vector data conversion on landscape indicies. *Canadian Journal of Forest Research*, *26*, 1416–1425.

Beyersbergen, G. W., Niemuth, N. D., & Norton, M. R. (2004). *Northern Prairie and Parkland Waterbird Conservation Plan*. *Northern Prairie & Parkland Waterbird Conservation Plan*. Denver Colorado.

Brown, P. H., & Lant, C. L. (1999). The effect of wetland mitigation banking on the achievement of no-net-loss. *Environmental Management*, *23*(3), 333–345. http://doi.org/10.1007/s002679900190

Bubar, C. J., McColl, S. J., & Hall, L. M. (2000). *Weeds of the Prairies*. Edmonton, Alberta: Alberta Agriculture and Rural Development.

Carver, S. J., & Brunsdon, C. F. (1994). Vector to raster conversion error and feature complexity: an empirical study using simulated data. *International Journal of Geographical Information Systems*, *8*(3), 261–270. http://doi.org/10.1080/02693799408901999

CEMA. (2014). *Guidelines for Wetlands Establishment*.

Chastant, J. E., Petersen, M. L., & Gawlik, D. E. (2017). Nesting substrate and water-level fluctuations influence wading bird nesting patterns in a large shallow eutrophic lake. *Hydrobiologia*, *788*(1), 371–383. http://doi.org/10.1007/s10750-016-3015-0

Choudhry, S., & Morad, M. (1998). GIS Errors and Surface Hydrologic Modeling: An Examination of Effects and Solutions. *Journal of Surveying Engineering*, *124*(3), 134. http://doi.org/http://dx.doi.org/10.1061/(ASCE)0733-9453(1998)124:3(134

Clare, S., & Creed, I. F. (2014). Tracking wetland loss to improve evidence-based wetland policy learning and decision making. *Wetlands Ecology and Management*, *22*(3). http://doi.org/10.1007/s11273-013-9326-2

Clare, S., Krogman, N., Foote, L., & Lemphers, N. (2011). Where is the avoidance in the implementation of wetland law and policy? *Wetlands Ecology and Management*, *19*(2), 165–182. http://doi.org/10.1007/s11273-011-9209-3

Congalton, R. G. (1997). Exploring and Evaluating the Consequences of Vector-to-Raster and Raster-to-Vector Conversion. *Photogrammetric Engineering & Remote Sensing*, *63*(April), 425–434.

Cowardin, L. M., Carter, V., Golet, F. C., & LaRoe, E. T. (1979). Classification of wetlands and deepwater habitats of the United States. *FGDC-STD-004-2013. Second Edition*, (December 1979), 79. http://doi.org/FWS/OBS-79/31

Dahl, T. E., & Watmough, M. D. (2007). Current approaches to wetland status and trends monitoring in prairie Canada and the continental United States of America. *CANADIAN JOURNAL OF REMOTE SENSING*, *33*, S17–S27.

Dale, V. H., & Beyeler, S. C. (2001). Challenges in the development and use of ecological indicators. *Ecological Indicators*, *1*(1), 3–10. http://doi.org/10.1016/S1470-160X(01)00003-6

Davidson, N. (2014). How much wetland has the world lost? Long-term and recent trends in global wetland area. *Marine and Freshwater Research*, *65*(10), 934–941. http://doi.org/https://doi.org/10.1071/MF14173

Dekeyser, E. S., Biondini, M., Kirby, D., & Hargiss, C. (2009). Low prairie plant communities of wetlands as a function of disturbance: Physical parameters. *Ecological Indicators*, *9*(2), 296–306. http://doi.org/10.1016/j.ecolind.2008.05.003

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

133. http://doi.org/10.1016/S1470-160X(03)00015-3

DiBiase, D. (2014). The Nature of Geographic Information. Retrieved from https://www.e-education.psu.edu/natureofgeoinfo/

Downing, D. J., & Pettapiece, W. W. (2006). *Natural Regions and Subregions of Alberta*. *Government of Alberta …*. http://doi.org/Pub. No. T/852

Esselman, P. C., Infante, D. M., Wang, L., Cooper, A. R., Wieferich, D., Tsang, Y. P., … Taylor, W. W. (2013). Regional fish community indicators of landscape disturbance to catchments of the conterminous United States. *Ecological Indicators*, *26*, 163–173. http://doi.org/10.1016/j.ecolind.2012.10.028

Euliss, N. H., Labaugh, J., & Fredrickson, L. (2004). The wetland continuum: A conceptual framework for interpreting biological studies, *24*(2).

Fay, P. A., Guntenspergen, G. R., Olker, J. H., & Johnson, W. C. (2016). Climate change impacts on freshwater wetland hydrology and vegetation cover cycling along a regional aridity gradient, *7*(October).

Fennessy, M. S., Jacobs, A. D., & Kentula, M. E. (2007). An evaluation of rapid methods for assessing the ecological condition of wetlands. *Wetlands*, *27*(3), 543–560. http://doi.org/10.1672/0277-5212(2007)27[543:AEORMF]2.0.CO;2

Forrest, A. (2010). *Created stormwater wetlands as wetland compensation and a floristic quality approach to wetland condition assessment in central Alberta*. *Biological Sciences*.

Gahegan, M., & Ehlers, M. (2000). A framework for the modelling of uncertainty between remote sensing and geographic information systems. *ISPRS Journal of Photogrammetry and Remote Sensing*, *55*(3), 176–188. http://doi.org/10.1016/S0924-2716(00)00018-6

Galatowitsch, S. M., & van der Valk, A. G. (1996). The Vegetation of Restored and Natural Prairie Wetlands. *Ecological Applications*, *6*(1), 102. http://doi.org/10.2307/2269557

Galatowitsch, S. M., Whited, D. C., Lehtinen, R., Husveth, J., & Schik, K. (2000). The vegetation of wet meadows in relation to their land-use. *ENVIRONMENTAL MONITORING AND ASSESSMENT*, *60*(2), 121–144. http://doi.org/10.1023/A:1006159028274

Gerard W. Beyersbergen, C. W. S., Neal D. Niemuth, U. S. F. and W. S., & Michael R. Norton, C. W. S.

(2004). Northern Prairie and Parkland Waterbird Conservation Plan. *Communication*, 1–57.

Gleason, J. E., & Rooney, R. C. (2017). Aquatic macroinvertebrates are poor indicators of agricultural activity in northern prairie pothole wetlands. *Ecological Indicators*, *81*(May), 333–339. http://doi.org/10.1016/j.ecolind.2017.06.013

Gleason, R. A., Euliss, N. H., Hubbard, D. E., & Duffy, W. G. (2003). Effects of sediment load on emergence of aquatic invertebrates and plants from wetland soil egg and seed banks. *Wetlands*, *23*(1), 26–34. http://doi.org/10.1672/0277-5212(2003)023[0026:EOSLOE]2.0.CO;2

Government of Alberta. (1993). *Wetland Management in the Settled Areas of Alberta: An Interim Policy*.

Government of Alberta. (2013). *Alberta Wetland Policy*. Edmonton, Alberta.

Government of Alberta. (2015). Alberta Conservation Information Management System (ACIMS).

Government of Alberta. (2016). *Alberta Wetland Restoration Directive*. *Water Policy Branch, Alberta Environment and Parks. Edmonton, Alberta.* Edmonton, Alberta.

Government of Canada. (1991). *The federal policy on wetland conservation*. Ottawa, Ontario.

Guareschi, S., Abellán, P., Laini, A., Green, A. J., Sánchez-Zapata, J. A., Velasco, J., & Millán, A. (2015). Cross-taxon congruence in wetlands: Assessing the value of waterbirds as surrogates of macroinvertebrate biodiversity in Mediterranean Ramsar sites. *Ecological Indicators*, *49*, 204–215. http://doi.org/10.1016/j.ecolind.2014.10.012

Hales, B. M., & Pronovost, P. J. (2006). The checklist-a tool for error management and performance improvement. *Journal of Critical Care*, *21*(3), 231–235. http://doi.org/10.1016/j.jcrc.2006.06.002

Hargis, C., Bissonette, J., & David, J. (1998). The behavior of landscape metrics commonly used in the study of habitat fragmentation. *Landscape Ecology*, *13*(1992), 167–186. http://doi.org/10.1023/a:1007965018633

Harrell, F. E., & Dupont, C. (2017). Hmisc: Harrell Miscellaneous R package version 4.0-3.

Hayashi, M., van der Kamp, G., & Rosenberry, D. O. (2016). Hydrology of Prairie Wetlands: Understanding the Integrated Surface-Water and Groundwater Processes. *Wetlands*, *36*, 237–254. http://doi.org/10.1007/s13157-016-0797-9

Hering, D., Feld, C. K., Moog, O., & Ofenbock, T. (2006). Cook book for the development of a Multimetric Index for biological condition of aquatic ecosystems: Experiences from the European AQEM and STAR projects and related initiatives. *Hydrobiologia*, *566*(1), 311–324. http://doi.org/10.1007/s10750-006-0087-2

Heywood, I., Cornelius, S., & Carver, S. (2006). *An Introduction To Geographical Information Systems Third Edition*.

Houlahan, J. E., Keddy, P. A., Makkay, K., & Findlay, C. S. (2006). The effects of adjacent land use on wetland species richness and community composition. *Wetlands*, *26*(1), 79–96. http://doi.org/10.1672/0277-5212(2006)26[79:TEOALU]2.0.CO;2

Hu, S., Niu, Z., Chen, Y., Li, L., & Zhang, H. (2017). Global wetlands: Potential distribution, wetland loss, and status. *Science of the Total Environment*, *586*, 319–327. http://doi.org/10.1016/j.scitotenv.2017.02.001

ITIS. (2016). Integrated Taxonomic Information System. Retrieved January 1, 2016, from http://www.itis.gov/

Jaeger, J. A. G. (2000). Landscape division, splitting index, and effective mesh size: New measures of landscape fragmentation. *Landscape Ecology*, *15*(2), 115–130. http://doi.org/10.1023/A:1008129329289

Kantrud, H. A., Millar, J. B., & van der Valk, A. G. (1989). Vegetation of wetlands of the prairie pothole region. In A. G. van der Valk (Ed.), *Northern prairie wetlands* (pp. 132–187). Ames, IA: Iowa State University Press.

Karr, J. R. (1981). Assessment of Biotic Integrity Using Fish Communities. *FISHERIES*, *6*(6), 21–27. http://doi.org/10.1577/1548-8446(1981)0062.0.CO;2

Karr, J. R. (2006). Seven Foundations of Biological Monitoring and Assessment. *Biologia Ambientale*, *20*(2), 7–18.

Karr, J. R., & Chu, E. W. (1999). *Restoring Life in Running Waters: Better Biological Monitoring*. Washington DC: Island Press.

Karr, J. R., & Chu, E. W. (2000). Sustaining living rivers. *HYDROBIOLOGIA*, *422*, 1–14. http://doi.org/10.1023/A:1017097611303

Karr, J. R., & Dudley, D. R. (1981). Ecological perspective on water quality goals. *Environmental Management*, *5*(1), 55–68. http://doi.org/10.1007/BF01866609

Keddy, P. (1992). Assembly and response rules: two goals for predictive community ecology. *Journal of Vegetation Science*, *3*(2), 157–164. http://doi.org/10.2307/3235676

Keddy, P. (1999). Wetland restoration: The potential for assembly rules in the service of conservation. *WETLANDS*, *19*(4), 716–732.

Keddy, P. (2000). *Wetland Ecology: Principles and Conservation*. Cambridge, UK: Cambridge University Press.

Kennedy, G., & Mayer, T. (2002). Natural and constructed wetlands in Canada: An overview. *WATER QUALITY RESEARCH JOURNAL OF CANADA*, *37*(2), 295–325.

Kershaw, L., Gould, J., Johnson, D., & Lancaster, J. (2001). *Rare Vascular Plants of Alberta*. Edmonton, Alberta: University of Alberta Press: Canadian Forest Service, Northern Forestry Center.

Kuehne, L. M., Olden, J. D., Strecker, A. L., Lawler, J. J., & Theobald, D. M. (2017). Past, present, and future of ecological integrity assessment for fresh waters. *Frontiers in Ecology and the Environment*, *15*(4), 197–205. http://doi.org/10.1002/fee.1483

Kuzovkina, Y. A., & Quigley, M. F. (2005). Willows beyond wetlands: Uses of salix L. species for environmental projects. *Water, Air, and Soil Pollution*, *162*(1–4), 183–204. http://doi.org/10.1007/s11270-005-6272-5

Landeiro, V. L., Bini, L. M., Costa, F. R. C., Franklin, E., Nogueira, A., De Souza, J. L. P., … Magnusson, W. E. (2012). How far can we go in simplifying biomonitoring assessments? An integrated analysis of taxonomic surrogacy, taxonomic sufficiency and numerical resolution in a megadiverse region. *Ecological Indicators*, *23*, 366–373. http://doi.org/10.1016/j.ecolind.2012.04.023

Leica CS10/CS15 & GS Sensors User Manual. (2014). Heerbrugg, Switzerland: Leica Geosystems.

Leitao, A. B., & Ahern, J. (2002). Applying landscape ecological concepts and metrics in sustainable landscape planning. *Landscape and Urban Planning*, *59*(2), 65–93. http://doi.org/10.1016/S0169-2046(02)00005-1

Liao, S., Bai, Z., & Bai, Y. (2012). Errors prediction for vector-to-raster conversion based on map load and cell size. *Chinese Geographical Science*, *22*(6), 695–704. http://doi.org/10.1007/s11769-012-0544-y

Lunde, K. B., & Resh, V. H. (2012). Development and validation of a macroinvertebrate index of biotic integrity (IBI) for assessing urban impacts to Northern California freshwater wetlands. *Environmental Monitoring and Assessment*, *184*(6), 3653–3674. http://doi.org/10.1007/s10661-011-2214-4

MacEachren, A. M., Robinson, A., Hopper, S., Gardner, S., Murray, R., Gahegan, M., & Hetzler, E. (2005). Visualizing geospatial information uncertainty: What we know and what we need to know. *… Information Science*, *32*(3), 139–160. http://doi.org/10.1559/1523040054738936

Mack, J. J. (2004). Integrated Wetland Assessment Program. Part 4: A vegetation index of biotic integrity (VIBI) and tiered aquatic life uses (TALUs) for Ohio wetlands. Ohio EPA technical report WET/2004-4., 1–91.

Mack, J. J. (2007). Developing a wetland IBI with statewide application after multiple testing iterations. *Ecological Indicators*, *7*(4), 864–881. http://doi.org/10.1016/j.ecolind.2006.11.002

Mazerolle, M. J. (2017). AICcmodavg: Model selection andmultimodel inference based on (Q)AIC(c). R package version 2.1-1.

McCauley, L. A., Anteau, M. J., Post van der Burg, M., & Wiltermuth, M. T. (2015). Land use and wetland drainage affect water levels and dynamics of remaining wetlands. *Ecosphere*, *6*(June), 1–22.

McCune, B., & Grace, J. B. (2002). *Analysis of Ecological Communities*. MjM Software Design.

McGarigal, K. (2015). Fragstats.Help.4.2, (April), 1–182. http://doi.org/10.1016/S0022-3913(12)00047-9

McGarigal, K., & Marks, B. J. (1994). FRAGSTATS: Spatial Pattern Analysis Program for Quantifying Landscape Structure. *General Technical Report PNW-GTR-351. US …*, *97331*(503), 134.

McKenzie, G., Hegarty, M., Barrett, T., & Goodchild, M. F. (2015). Assessing the effectiveness of different visualizations for judgments of positional uncertainty. *International Journal of Geographical Information Science*, *30*(2), 221–239. http://doi.org/10.1080/13658816.2015.1082566

Meador, M. R., Whittier, T. R., Goldstein, R. M., Hughes, R. M., & Peck, D. V. (2008). Evaluation of an Index of Biotic Integrity Approach Used to Assess Biological Condition in Western U.S. Streams and Rivers at Varying Spatial Scales. *Transactions of the American Fisheries Society*, *137*(1), 13–22. http://doi.org/10.1577/T07-054.1

Miller, K. M., Mitchell, B. R., & McGill, B. J. (2016). Constructing multimetric indices and testing ability of landscape metrics to assess condition of freshwater wetlands in the Northeastern US. *Ecological Indicators*, *66*, 143–152. http://doi.org/10.1016/j.ecolind.2016.01.017

Miranda, L. E., Aycock, J. N., & Killgore, K. J. (2012). A direct-gradient multivariate index of biotic condition. *Transactions of the American Fisheries Society*, *141*(6), 1637–1648. http://doi.org/10.1080/00028487.2012.717519

Mitsch, W. J., & Gosselink, J. G. (2007). *Wetlands* (Fourth). Hoboken, New Jersey: JOHN WILEY & SONS Inc.

Moreno-Mateos, D., Power, M. E., Comín, F. A., & Yockteng, R. (2012). Structural and functional loss in restored wetland ecosystems. *PLoS Biology*, *10*(1), e1001247. http://doi.org/10.1371/journal.pbio.1001247

Moss, E. H. (Ezra H., & Packer, J. G. (1983). *Flora of Alberta : a manual of flowering plants, conifers, ferns and fern allies found growing without cultivation in the province of Alberta, Canada*. University of Toronto Press.

Naugle, D. E., Higgins, K. E., Nusser, S. M., & Johnson, W. C. (1999). Scale-dependent habitat use in three species of prairie wetland birds. *Landscape Ecology*, *14*, 267–276. http://doi.org/10.1023/A:1008088429081.

NGPFQA. (2001). Coefficients of conservatism for the vascular flora of the Dakotas and adjacent grasslands. *Information and Technology Report*.

Odland, A., & del Moral, R. (2002). Thirteen years of wetland vegetation succession following a permanent drawdown, Myrkdalen Lake, Norway. *Plant Ecology*, *162*(2), 185–198. http://doi.org/10.1023/A:1020388910724

Olechnowski, B. F. M., & Debinski, D. M. (2008). Response of songbirds to riparian willow habitat structure in the Greater Yellowstone Ecosystem. *The Wilson Journal of Ornithology*, *120*(4), 830–839. http://doi.org/10.1676/07-132.1

Ontario Ministry of Natural Resources. (2013). *Ontario Wetland Evaluation System*.

Paradeis, B. L., DeKeyser, E. S., & Kirby, D. R. (2010). Evaluation of Restored and Native Prairie Pothole Region Plant Communities Following an Environmental Gradient. *Natural Areas Journal*, *30*(3), 294–304. http://doi.org/10.3375/043.030.0305

Pipp, A. (2015). *Coefficient of conservatism rankings for the flora of Montana.* Helena, MT : Montana Natural Heritage Program,.

Poiani, K. A., & Johnson, W. C. (2012). A Spatial Simulation Model of Hydrology and Vegetation Dynamics in Semi-Permanent Prairie Wetlands, *3*(2), 279–293.

Puchniak Begley, A. J., Gray, B. T., & Paszkowski, C. A. (2012). A comparison of restored and natural wetlands as habitat for birds in the Prairie Pothole Region of Saskatchewan, Canada. *Raffles Bulletin of Zoology*, (SUPPL.25), 173–187.

Raab, D., & Bayley, S. E. (2012). A vegetation-based Index of Biotic Integrity to assess marsh reclamation success in the Alberta oil sands, Canada. *Ecological Indicators*, *15*(1), 43–51. http://doi.org/10.1016/j.ecolind.2011.09.025

R Core Team. (2017). R: A language and environment for statistical computing. Vienna, Austria: R Foundation for Statistical Computing.

Ramsar Convention Secretariat. (2010). National Wetland Policies: Developing and Implementing National Wetland Policies. *Ramsar Handbooks for the Wise Use of Wetlands*, *2*, 64.

Reynoldson, T. B., Norris, R. H., Resh, V. H., Day, K. E., & Rosenberg, D. M. (1997). The Reference Condition : A Comparison of Multimetric and Multivariate Approaches to Assess Water-Quality Impairment Using Benthic Macroinvertebrates. *Journal of the North American Benthological Society*, *16*(4), 833–852.

Riffell, S. K., Keas, B. E., & Burton, T. M. (2001). Area and habitat relationships of birds in Great Lakes coastal wet meadows. *Wetlands*, *21*(4), 492–507. http://doi.org/10.1672/0277-5212(2001)021[0492:AAHROB]2.0.CO;2

Riffell, S. K., Keas, B. E., & Burton, T. M. (2003). Birds in North American Great lakes coastal wet meadows: is landscape context important? *Landscape Ecology*, *:18*(1995), 95–111.

Rooney, R. C., & Bayley, S. E. (2010). Quantifying a stress gradient: An objective approach to variable selection, standardization and weighting in ecosystem assessment. *Ecological Indicators*, *10*(6), 1174–1183. http://doi.org/10.1016/j.ecolind.2010.04.001

Rooney, R. C., & Bayley, S. E. (2011). Development and testing of an index of biotic integrity based on submersed and floating vegetation and its application to assess reclamation wetlands in Alberta's oil sands area, Canada. *Environmental Monitoring and Assessment*, *184*(2), 749–761. http://doi.org/10.1007/s10661-011-1999-5

Rooney, R. C., & Bayley, S. E. (2012a). Community congruence of plants, invertebrates and birds in natural and constructed shallow open-water wetlands: Do we need to monitor multiple assemblages? *Ecological Indicators*, *20*, 42–50. http://doi.org/10.1016/j.ecolind.2011.11.029

Rooney, R. C., & Bayley, S. E. (2012b). Development and testing of an index of biotic integrity based on submersed and floating vegetation and its application to assess reclamation wetlands in Alberta's oil sands area, Canada. *Environmental Monitoring and Assessment*, *184*(2), 749–61. http://doi.org/10.1007/s10661-011-1999-5

Russi, D., ten Brink, P., Farmer, a., Badura, T., Coates, D., Förster, J., … Russi, Daniela; ten Brink, Patrick; Farmer, Andrew; Badura, Tomas; Coates, David; Förster, Johannes; Kumar, Ritesh; Davidson, N. (2013). The Economics of Ecosystems and Biodiversity for Water and Wetlands. *Vasa*, 84. http://doi.org/10.1007/s13398-014-0173-7.2

Schindler, D. W., & Donahue, W. F. (2006). An impending water crisis in Canada's western prairie provinces. *Proceedings of the National Academy of Sciences of the United States of America*, *103*(19), 7210–6. http://doi.org/10.1073/pnas.0601568103

Schneider, R. R. (2013). *Alberta's Natural Subregions Under a Changing Climate: Past, Present, and Future*. Edmonton, Alberta.

Schoolmaster, D. R., Grace, J. B., & Schweiger, E. W. (2012). A general theory of multimetric indices and their properties. *Methods in Ecology and Evolution*, *3*(4), 773–781. http://doi.org/10.1111/j.2041-210X.2012.00200.x

Sculthorpe, C. D. (1967). *The Biology of Aquatic Vascular Plants.* (Vol. 53). London: Edward Arnold Ltd. http://doi.org/10.1002/iroh.19680530207

Seabloom, E. W., Moloney, K. A., & van der Valk, A. G. (2001). Constraints on the establishment of plants along a fluctuating water-depth gradient. *Ecology*, *82*(8), 2216–2232. http://doi.org/10.1890/0012-9658(2001)082[2216:COTEOP]2.0.CO;2

Seabloom, E. W., & van der Valk, A. G. (2003a). Plant diversity, composition, and invasion of restored and natural prairie pothole wetlands: Implications for restoration. *Wetlands*, *23*(1), 1–12. http://doi.org/10.1672/0277-5212(2003)023[0001:PDCAIO]2.0.CO;2

Seabloom, E. W., & van der Valk, A. G. (2003b). The development of vegetative zonation patterns in restored prairie pothole wetlands. *Journal of Applied Ecology*, *40*(1), 92–100. http://doi.org/10.1046/j.1365-2664.2003.00764.x

Serran, J., & Creed, I. (2014). *The Preferential Loss of Small Geographically Isolated Wetlands on Prairie Landscapes*.

Serran, J., & Creed, I. (2015). New mapping techniques to estimate the preferential loss of small wetlands on prairie landscapes. *Hydrological Processes*, n/a-n/a. http://doi.org/10.1002/hyp.10582

Shipley, B. (2010). *From Plant Traits to Vegetation Structure: Chance and Selection in the Assembly of Ecological Communities*. Cambridge University Press.

Sırrı Mara, S., Hadi Mara, H., Aktu, B., Emin Mara, E., & Yildiz, F. (2010). Topological error correction of GIS vector data. *International Journal of the Physical Sciences*, *5*(5), 476–483.

Stevens, J., Smith, J. M., & Biancheti, R. A. (2012). Mapping our Changing World. Retrieved September 24, 2016, from https://www.e-education.psu.edu/geog160/node/1925

Stewart, R. E., & Kantrud, H. A. (1971). *Classification of Natural Ponds and Lakes in the Glaciated Prairie Region*.

Stewart, R. E., & Kantrud, H. A. (1972). *Vegetation of prairie potholes, North Dakota, in relation to quality of water and other environmental factors*. *Geological Survey Professional Paper* (Vol. 585–D).

Stoddard, J. L., Herlihy, A. T., Peck, D. V., Hughes, R. M., Whittier, T. R., & Tarquinio, E. (2008). A process for creating multimetric indices for large-scale aquatic surveys. *Journal of the North American Benthological Society*, *27*(4), 878–891. http://doi.org/10.1899/08-053.1

Stoddard, J. L., Larsen, D. P., Hawkins, C. P., Johnson, R. K., & Norris, R. H. (2006). Setting expectations for the ecological condition of streams: The concept of reference condition. *Ecological Applications*, *16*(4), 1267–1276.

SX Blue II+ GNSS Technical Schematics. (2015). Montreal: Geneq Inc.

Symonds, M. R. E., & Moussalli, A. (2011). A brief guide to model selection, multimodel inference and model averaging in behavioural ecology using Akaike information criterion. *Behav Ecol Sociobiol*, *65*(1), 13–21. http://doi.org/10.1007/s00265-010-1037-6

Taft, J., Wilhelm, G., Ladd, D., & Masters, L. (1997). Floristic quality assessment for vegetation in Illinois: A method for assessing vegetation integrity. *Erigenia*, (1977), 3–95.

Timoney, K. P., & Argus, G. (2006). Willows, water regime, and recent cover change in the Peace–Athabasca Delta. *Écoscience*, *13*(3), 308–317. http://doi.org/10.2980/i1195-6860-13-3-308.1

Tiner, R. W. (1999). *Wetland Indicators: A guide to Wetland Identification, Delineation, Classification, and Mapping*. CRC PRESS-TAYLOR & FRANCIS GROUP, 6000 BROKEN SOUND PARKWAY NW, STE 300, BOCA RATON, FL 33487-2742 USA.

Turner, M. G. (1989). Landscape Ecology: The Effect of Pattern on Process. *Annual Review of Ecology and Systematics*, *20*(1), 171–197. http://doi.org/10.1146/annurev.es.20.110189.001131

U.S. Army Corps of Engineers. (1987). *Corps of Engineers Wetlands Delineation Manual*. *Wetlands Research Program Technical Report Y-87-1*. Vicksburg, MS.

U.S.DA. (2017a). Fire Effects Informatin System (FEIS). Retrieved January 1, 2016, from https://www.feis-crs.org/feis/

U.S.DA. (2017b). United States Department of Agriculture Natural Resources Conservation Services. Retrieved from https://plants.usda.gov/java/

U.S.EPA. (2002a). *Methods for Evaluating Wetland Condition #10: Using Vegetation To Assess Environmental Conditions in Wetlands* (Vol. 11).

U.S.EPA. (2002b). *Methods for Evaluating wetland condition #6: Developing Metrics and Indexes of Biological Integrity* (Vol. EPA-822-R-). Washington, DC.

U.S.EPA. (2017). Wetlands Monitoring and Assessment. Retrieved December 29, 2017, from https://www.epa.gov/wetlands/wetlands-monitoring-and-assessment#regional

Ubeda, T., & Egenhofer, M. J. (1997). Topological Error Correcting in GIS. In *Advances in Spatial Databases -- Fifth International Symposium on Large Spatial Databases*.

van der Kamp, G., & Hayashi, M. (2008). Groundwater-wetland ecosystem interaction in the semiarid glaciated plains of North America. *Hydrogeology Journal*, *17*(1), 203–214. http://doi.org/10.1007/s10040-008-0367-1

van der Kamp, G., Hayashi, M., & Gallén, D. (2003). Comparing the hydrology of grassed and cultivated catchments in the semi-arid Canadian prairies. *Hydrological Processes*, *17*(3), 559–575. http://doi.org/10.1002/hyp.1157 van der Valk, A. G. (1981). Succession in Wetlands : A Gleasonian Approach. *Ecological Soceity of America*, *62*(3), 688–696.

van der Valk, A. G. (2005). Water-level fluctuations in North American prairie wetlands. *Hydrobiologia*, *539*(1), 171–188. http://doi.org/10.1007/s10750-004-4866-3

van der Valk, A. G., & Pederson, R. L. (2003). The swancc decision and its implications for prairie potholes. *Wetlands*, *23*(3), 590–596. http://doi.org/10.1672/0277-5212(2003)023[0590:TSDAII]2.0.CO;2

Van Leeuwen, W. J. D., & Orr, B. J. (2006). Spectral vegetation indices and uncertainty: Insights from a user's perspective. *IEEE Transactions on Geoscience and Remote Sensing*, *44*(7), 1931–1933. http://doi.org/10.1109/TGRS.2006.873688

van Meter, K. J., & Basu, N. B. (2015). Signatures of human impact: size distributions and spatial organization of wetlands in the Prairie Pothole landscape. *Ecological Applications*, *25*(2), 451–465. http://doi.org/10.1890/14-0662.1

van Sickle, J. (2010). Correlated Metrics Yield Multimetric Indices with Inferior Performance. *Transactions of the American Fisheries Society*, *139*(January 2014), 1802–1817. http://doi.org/10.1577/T09-204.1

Verhoeven, J. T. A., Soons, M. B., Janssen, R., & Omtzigt, N. (2008). An Operational Landscape Unit approach for identifying key landscape connections in wetland restoration. *Journal of Applied Ecology*, *45*(5), 1496–1503. http://doi.org/10.1111/j.1365-2664.2008.01534.x

Voc, C. C., Verboom, J., Opdam, P. F. M., & Ter Braak, C. J. F. (2001). Toward ecologically scaled landscape indicies. *The American Naturalist*, *183*(1), 284–290.

Wade, T. G., Wickham, J. D., Nash, M. S., Neale, a C., Riitters, K. H., & Jones, K. B. (2003). A Comparison of Vector and Raster GIS Methods for Calculating Landscape Metrics Used in Environmental Assessments. *Photogrammetric Engineering and Remote Sensing*, *69*(12), 1399–1405. http://doi.org/10.14358/PERS.69.12.1399

Welling, C. H., Pederson, R. L., & van der Valk, A. G. (2012). Recruitment from the Seed Bank and the Development of Zonation of Emergent Vegetation During a Drawdown in a Prairie Wetland, *76*(2), 483–496.

Werner, K. J., & Zedler, J. B. (2002). How sedge meadow soils, microtopography, and vegetation respond to sedimentation. *Wetlands*, *22*(3), 451–466. http://doi.org/10.1672/0277-

5212(2002)022[0451:HSMSMA]2.0.CO;2

Willby, N. J., Pulford, I. D., & Flowers, T. H. (2001). Tissue nutrient signatures predict herbaceous-wetland community responses to nutrient availability. *New Phytologist*, *152*(3), 463–481. http://doi.org/10.1046/j.0028-646X.2001.00274.x

Wilson, M. J., & Bayley, S. E. (2012). Use of single versus multiple biotic communities as indicators of biological integrity in northern prairie wetlands. *Ecological Indicators*, *20*, 187–195. http://doi.org/10.1016/j.ecolind.2012.02.009

Wilson, M. J., Bayley, S. E., & Rooney, R. C. (2013). A plant-based index of biological integrity in permanent marsh wetlands yields consistent scores in dry and wet years. *Aquatic Conservation: Marine and Freshwater Ecosystems*, *23*(5), n/a-n/a. http://doi.org/10.1002/aqc.2354

Wilson, M. J., Forrest, A. S., & Bayley, S. E. (2013). Floristic quality assessment for marshes in Alberta's northern prairie and boreal regions. *AQUATIC ECOSYSTEM HEALTH & MANAGEMENT*, *16*(3), 288–299. http://doi.org/10.1080/14634988.2013.825194

Wormley, S. (2010). GPS Error & Estimating your receiver's accuracy. Retrieved from http://www.edu-observatory.org/gps/gps_accuracy.html

Wright, C. K., & Wimberly, M. C. (2013). Recent land use change in the Western Corn Belt threatens grasslands and wetlands. *Proceedings of the National Academy of Sciences*, *110*(10), 4134–4139. http://doi.org/10.1073/pnas.1215404110

Wu, J. (2006). Landscape ecology, cross-disciplinarity, and sustainability science. *Landscape Ecology*, *21*(1), 1–4. http://doi.org/10.1007/s10980-006-7195-2

Zoltai, S. C., & Vitt, D. H. (1995). Canadian wetlands: Environmental gradients and classification. *Vegetatio*, *118*(1–2), 131–137. http://doi.org/10.1007/BF00045195

Appendices

Appendix 2.1. Disturbance Score Calculation from Anderson (2017).

MMI development requires an objective basis for ranking wetlands. For my study region, there were no existing quantitative or qualitative tools to rank wetlands from the least to most disturbed condition. I created a qualitative disturbance index that used the extent of non-natural land cover around each wetland as the basis for determining wetland condition. I determined that the extent of non-natural disturbance within a 500 m buffer did not adequately characterize the non-natural disturbance at a site, as within wetland disturbances also influenced wetland condition, but were not evident from surrounding land cover. To represent within wetland non-natural disturbances, I included modifiers in my index to build upon the disturbances characterized in the 500 m buffer around each wetland. The within wetland disturbance modifiers that I included were the presence of cattle disturbance, soil pesticides, and within wetland agricultural activity. The modifiers I included in my disturbance index were common categories used in existing qualitative, rapid assessment tools (Fennessy et al., 2007; Mack, 2007).

The disturbance scores are based on the % non-natural land cover within a 500 m buffer, for example, if a site had 38 % non-natural cover within the buffer, the wetland was assigned 38 points. Additional modifiers are then applied that may raise the score of the site. If cattle disturbance was detected within the delineated wetland boundary, it was determined to be either low or high intensity based on technician field notes and assigned points accordingly, +0 for no grazing, +25 points for low intensity or +50 points for high grazing intensity. Sediment samples that I collected in August were analyzed for a comprehensive list of pesticides by Dr. Claudia Sheedy at the Lethbridge Agriculture and Agri-Food Canada pesticide lab (see below). For my index, I added 50 points if any pesticides were detected in the sediment; however, I excluded legacy pesticide compounds that may reflect historic land use that no longer influences wetland vegetation. Thus, I excluded any non-registered or delisted pesticides that were detected in wetland sediment. For the last modifier, if any agricultural activity was evident within the delineated wetland boundary, I added 50 points to the disturbance score. The total possible disturbance index score was thus 250 points (100% surrounding agriculture + 50 points for evidence of high intensity grazing + 50 points for the presence of pesticides in wetland sediment + 50 points for having agricultural activities take place within the wetland boundary). Thus, higher disturbance scores representing sites with higher levels of non-natural disturbance.

Example calculation:

Site 117

Disturbance Index Scoring Criteria	**Site Information**	**Score**
Percent non-natural land cover in 500 m buffer around wetland	91 %	91
Cattle disturbance	None	0
Sediment pesticides	Present	50
Buffer: Agricultural activity within wetland	Absent	0
		141

Disturbance score: 141

List of pesticide compounds that were analyzed for in wetland sediment samples. Only registered pesticides included.

2,4-Dichlorophenoxyacetic acid
2,4-Dichlorophenol
Azoxystrobin
Bentazon
Bromoxynil
Boscalid
Chlorothalonil
Chlorpyrifos
Clopyralid
Diazinon
Diclofop
Difenoconazole
Ethalfluralin
Fenoxaprop
Fluroxypyr
Imazamethabenz
Iprodione
MCPA (2-methyl-4-chlorophenoxyacetic acid)
Propiconazole
Propoxur
Prothioconazole-Desthio
Quizalofop-ethyl
Tebuconazole
Triallate
Trifluralin
Triticonazole

Appendix 2.2 Site characteristics used to calculate disturbance scores (as described in Appendix 2.1), final disturbance scores used in MMI development and validation for all MMI's from Anderson (2017).

Site ID	Non-natural Cover (%)	Grazing Intensity (0-none, 1-Low, 2-High)	Sediment Pesticides (without legacies)	Buffer (Agriculture in wetland 0-buffer, 1-no buffer)	Disturbance Score	Validation Dataset
98	0	1	0	1	75	
101	0	2	1	1	150	
109	86	2	1	1	236	
110	22	2	1	1	172	
115	97	1	1	1	222	
117	91	0	1	0	141	
124	0	1	1	1	125	X
131	0	2	1	1	150	
133	14	1	0	1	89	
135	29	1	1	1	154	
142	0	1	1	1	125	
145	95	1	0	1	170	
149	86	0	1	1	186	X
152	5	1	1	1	130	X
153	5	0	1	0	55	
158	0	1	0	1	75	
165	0	2	0	1	100	
173	69	1	0	1	144	X
184	100	0	1	1	200	
186	0	2	1	1	150	
188	73	0	1	1	173	
202	23	1	0	1	98	
203	100	2	0	1	200	X
308	69	1	0	1	144	
312	95	0	0	1	145	X

Site ID	Non-natural Cover (%)	Grazing Intensity (0-none, 1-Low, 2-High)	Sediment Pesticides (without legacies)	Buffer (Agriculture in wetland 0-buffer, 1-no buffer)	Disturbance Score	Validation Dataset
336	0	1	0	1	75	X
338	27	1	0	1	102	X
345	78	1	0	1	153	X
346	41	1	1	1	166	
360	66	2	1	1	216	
366	43	0	0	1	93	
375	3	2	1	1	153	
379	4	2	1	1	154	
384	88	1	0	1	163	
388	3	1	1	1	128	
KIN	10	1	1	1	135	
10	79	2	1	1	229	
13	44	2	0	1	144	
18	82	0	1	1	182	X
25	80	0	1	1	180	X
30	96	0	1	1	196	
31	62	0	0	1	112	X
32	13	1	0	1	88	
35	45	2	1	1	195	
56	98	0	1	1	195	X
67	27	0	1	0	77	X
89	94	0	1	1	194	X
90	100	0	0	1	150	X
182	99	0	1	1	199	
187	76	0	1	1	176	
190	92	0	1	0	142	X
194	36	1	1	1	161	

Site ID	Non-natural Cover (%)	Grazing Intensity (0-none, 1-Low, 2-High)	Sediment Pesticides (without legacies)	Buffer (Agriculture in wetland 0-buffer, 1-no buffer)	Disturbance Score	Validation Dataset
195	8	2	0	1	108	
200	46	0	1	0	96	
301	85	0	1	1	185	
317	57	2	0	1	157	
321	93	2	1	1	243	
333	13	1	0	1	88	
344	81	0	0	0	81	
351	71	0	1	0	121	X
365	26	0	0	0	26	
368	13	2	0	1	113	X
377	71	0	1	0	121	
395	29	0	0	0	29	
396	29	0	0	0	29	X
398	65	0	0	0	65	
BATL	5	0	1	0	55	X
GAD	19	0	0	0	19	
JJCOLL	15	0	0	0	15	X
MIQ	3	0	0	0	3	
RUM	0	1	0	1	75	
TOL	8	1	1	1	133	X

Appendix 2.3 Blank field sampling sheet with Braun-Blanquet cover classes. % Cover classes include: <0.01, 0.01-0.25, 0.25-1, 1-2.5, 2.5-5, 5-7.5, 7.5-10, 10-15, 15-25, 25-33, 33-50, 50-66, 66-75, 75-100%.

Vegetation % cover data sheet 2014

Site: ______________________ Personnel: ____________________ Date:___________________

Polygon name: **Polygon Area:** m^2

Species name	**Coll #**	**Q#:** **GPS:** **Robel:**	**Q#:** **GPS:** **Robel:**	**Q#:** **GPS:** **Robel:**	**Q#:** **GPS:** **Robel:**	**Q#:** **GPS:** **Robel:**

Appendix 2.4 Plant traits

Table A. Plant traits and their sources as determined by Adam Kraft.

Parameter	Explanation	Source
Code	Unique 8-character code, comprising (in most cases) the first 3 letters of the genus and the first 5 letters of the specific epithet	ITIS, 2016
Genus	Generic ranking	ITIS, 2016
Specific Epithet	Specific ranking	ITIS, 2016
Family	Familial ranking	ITIS, 2016
Group	Broad taxonomic ranking of a plant. SV = seedless vascular; M = monocot; E = eudicot	ITIS, 2016
GL 2014	If the plant was found in 2014 in the Grassland Natural Region	Field Observations
PL 2014	If the plant was found in 2014 in the Parkland Natural Region	Field Observations
GL 2015	If the plant was found in 2015 in the Grassland Natural Region	Field Observations
PL 2015	If the plant was found in 2015 in the Parkland Natural Region	Field Observations
Native Status	Whether a plant is indigenous to Alberta. N = native; I = introduced	Moss & Packer, 1983
Exotic?	Whether a plant has been identified as being exotic to Alberta according to the ACIMS; E = exotic, N = native	ACIMS, 2015
S Rank	Sub-national level NatureServe ranking; applicable to species only	ACIMS, 2015
G Rank	Global-level NatureServe ranking; applicable to species only	ACIMS, 2015
Watched	Whether a plant is currently watched or tracked by the Alberta Conservation Information Management System (ACIMS)	Government of Alberta, 2015
Rare	Whether a plant has been identified as being rare in Alberta; R = rare	Kershaw et al., 2001
Weed	Whether a plant has been identified as a weed in the Canadian Prairie Provinces; W = weed	Bubar et al., 2000
Noxious	Whether a plant is on the Noxious Weeds list in Alberta, as determined by the Alberta Weed Regulatory Advisory Council (AWRAC); N = noxious	AWRAC, 2014
CC Score (GL)	Coefficient of Conservatism score for the Grassland ecoregion (0-10). Exotic plants (according to ACIMS) were assigned a score of 0. Scores were first assigned according to Pipp 2015, then by Forrest 2010, then by NGPFQAP 2001.	Forrest, 2010; NGPFQA, 2001; Pipp, 2015
CC Score (PL)	Coefficient of Conservatism score for the Parkland ecoregion (0-10). Exotic plants (according to ACIMS), were assigned a score of 0. Scores were first assigned according to Forrest 2010, then by Pipp 2015, then by NGPFQAP 2001.	Forrest, 2010; NGPFQA, 2001; Pipp, 2015

Parameter	Explanation	Source
Sensitivity/ Tolerance (GL)	Whether a plant is sensitive to disturbance (CC 6-10) or tolerant (CC 0-2) according to its regional CC score	Andreas et al., 2004
Sensitivity/ Tolerance (PL)	Whether a plant is sensitive to disturbance (CC 6-10) or tolerant (CC 0-2) according to its regional CC score	Andreas et al., 2004
Lifecycle	Lifecycle strategy of a plant. P = perennial; B = biennial; A = annual	Moss & Packer, 1983
Wetland Status	Wetland plant indicator status, determined by the United States Department of Agriculture for the Great Plains Region	U.S.DA, 2017
Habit	Growth form of a plant. T = tree; S = shrub; F = herbaceous forb; V = vine-like forb; G = graminoid; A = aquatic	Moss & Packer, 1983
Form	Wetland vegetation form, sensu Ontario Wetland Evaluation System (OWES); H = hardwood trees; TS = tall shrubs (1-6m); LS = low shrubs (<1m); GC = groundcover/non-emergent forbs; NE = narrow-leaved emergent graminoids; RE = robust emergent graminoids; BE = broad-leaved emergent forbs; F = floating-leaved plants; FF = free-floating plants; SU = submerged plants	Ontario Ministry of Natural Resources, 2013
Vegetative Reproduction	Whether the plant is able to reproduce by vegetative means. R = rhizomes; S = stolons or trailing/rooting stems; T = turions	Moss & Packer, 1983
Nitrogen-Fixing	Whether a plant is capable of nitrogen fixation	Moss & Packer, 1983
Recalcitrant Litter	Whether plant litter is slow to decompose	U.S.EPA, 2002b
Dispersal Mechanism	The main means of propagule dispersal; A = anemochory (wind); H = hydrochory (water); Z = zoochory (animal)	Sculthorpe, 1967; U.S.DA, 2017a

Table B: Plant species and their taxonomy, regional detection and status. Note that GL stands for Grassland and PL stands for Parkland. ACIMS is the Alberta Conservation Information Management System maintained by Alberta Environment and Parks.

	Taxonomy				Detection				Status	
Code	Genus	Specific Epithet	Family	Group	GL 2014	PL 2014	GL 2015	PL 2015	Native Status	Exotic ACIMS
ACHALPIN	*Achillea*	*alpina*	Asteraceae	E		1			N	N
ACHMILLE	*Achillea*	*millefolium*	Asteraceae	E	1	1	1	1	N	N
ACOCALAM	*Acorus*	*calamus*	Acoraceae	M		1		1	N	N
AGRCRIST	*Agropyron*	*cristatum*	Poaceae	M			1		I	E
AGRGIGAN	*Agrostis*	*gigantea*	Poaceae	M				1	I	E
AGRSCABR	*Agrostis*	*scabra*	Poaceae	M	1	1	1	1	N	N
AGRSTRIA	*Agrimonia*	*striata*	Rosaceae	E		1		1	N	N
ALITRIVI	*Alisma*	*triviale*	Alismataceae	M	1	1	1	1	N	N
ALOAEQUA	*Alopecurus*	*aequalis*	Poaceae	M	1	1	1	1	N	N
ALOPRATE	*Alopecurus*	*pratensis*	Poaceae	M					I	E
AMARETRO	*Amaranthus*	*retroflexus*	Amaranthaceae	E	1		1		I	E
AMEALNIF	*Amelanchier*	*alnifolia*	Rosaceae	E					N	N
ANAMINIM	*Anagalilis*	*minima*	Primulaceae	E	1				N	N
ANECANAD	*Anemone*	*canadensis*	Ranunculaceae	E	1		1		N	N
ANTPARVI	*Antennaria*	*parvifolia*	Asteraceae	E		1			N	N
ARNCHAMI	*Arnica*	*chamissonis*	Asteraceae	E			1		N	N
ARTBIENN	*Artemisia*	*biennis*	Asteraceae	E	1	1	1	1	N	N
ARTCAMPE	*Artemisia*	*campestris*	Asteraceae	E			1		N	N
ARTLONGI	*Artemisia*	*longifolia*	Asteraceae	E	1	1	1		N	N
ARTLUDOV	*Artemisia*	*ludoviciana*	Asteraceae	E	1		1	1	N	N
ATRPROST	*Atriplex*	*prostrata*	Amaranthaceae	E	1		1		I	E
AVEFATUA	*Avena*	*fatua*	Poaceae	M	1				I	E
BECSYZIG	*Beckmannia*	*syzigachne*	Poaceae	M	1	1	1	1	N	N
BIDCERNU	*Bidens*	*cernua*	Asteraceae	E	1	1		1	N	N
BOLMARIT	*Bolboschoenus*	*maritimus*	Cyperaceae	M					N	N
BRANAPUS	*Brassica*	*napus*	Brassicaceae	E	1				I	E

	Taxonomy				Detection				Status	
Code	Genus	Specific Epithet	Family	Group	GL 2014	PL 2014	GL 2015	PL 2015	Native Status	Exotic ACIMS
BROINERM	*Bromus*	*inermis*	Poaceae	M	1	1	1		N	E
CALCANAD	*Calamagrostis*	*canadensis*	Poaceae	M	1	1		1	N	N
CALIPALU	*Callitriche*	*palustris*	Plantaginaceae	E	1	1			N	N
CALLPALU	*Calla*	*palustris*	Araceae	M		1			N	N
CALSTRIC	*Calamagrostis*	*stricta*	Poaceae	M	1	1	1	1	N	N
CALTPALU	*Caltha*	*palustris*	Ranunculaceae	E		1			N	N
CAPBURSA	*Capsella*	*bursa-pastoris*	Brassicaceae	E	1			1	I	E
CARAQUAT	*Carex*	*aquatilis*	Cyperaceae	M	1	1			N	N
CARATHER	*Carex*	*atherodes*	Cyperaceae	M	1	1	1	1	N	N
CARATHRO	*Carex*	*athrostachya*	Cyperaceae	M			1	1	N	N
CARBEBBI	*Carex*	*bebbii*	Cyperaceae	M	1	1		1	N	N
CARBREVI	*Carex*	*brevior*	Cyperaceae	M			1		N	N
CARCARVI	*Carum*	*carvi*	Apiaceae	E		1			I	E
CARDIAND	*Carex*	*diandra*	Cyperaceae	M					N	N
CARLACUS	*Carex*	*lacustris*	Cyperaceae	M		1			N	N
CARPELLI	*Carex*	*pellita*	Cyperaceae	M	1	1	1	1	N	N
CARPRAEG	*Carex*	*praegracilis*	Cyperaceae	M	1	1			N	N
CARPRATI	*Carex*	*praticola*	Cyperaceae	M					N	N
CARRETRO	*Carex*	*retrorsa*	Cyperaceae	M	1				N	N
CARSARTW	*Carex*	*sartwellii*	Cyperaceae	M					N	N
CARSYCHN	*Carex*	*sychnocephala*	Cyperaceae	M		1			N	N
CARUTRIC	*Carex*	*utriculata*	Cyperaceae	M	1	1		1	N	N
CERARVEN	*Cerastium*	*arvense*	Caryophyllaceae	E		1			N	N
CHAANGUS	*Chamerion*	*angustifolium*	Onagraceae	E		1			N	N
CHEALBUM	*Chenopodium*	*album*	Amaranthaceae	E	1	1	1	1	I	E
CHECAPIT	*Chenopodium*	*capitatum*	Amaranthaceae	E	1	1			N	N
CHERUBRU	*Chenopodium*	*rubrum*	Amaranthaceae	E				1	N	N
CICMACUL	*Cicuta*	*maculata*	Apiaceae	E		1		1	N	N

Taxonomy					Detection				Status	
Code	Genus	Specific Epithet	Family	Group	GL 2014	PL 2014	GL 2015	PL 2015	Native Status	Exotic ACIMS
CIRARVEN	*Cirsium*	*arvense*	Asteraceae	E	1	1	1	1	I	E
CIRVULGA	*Cirsium*	*vulgare*	Asteraceae	E	1		1		I	E
COLLINEA	*Collomia*	*linearis*	Polemoniaceae	E	1		1		N	N
COMPALUS	*Comarum*	*palustre*	Rosaceae	E		1			N	N
CORSERIC	*Cornus*	*sericea*	Cornaceae	E		1			N	N
CRETECTO	*Crepis*	*tectorum*	Asteraceae	E	1		1		I	E
DESCESPI	*Deschampsia*	*cespitosa*	Poaceae	M	1	1	1		N	N
DESSOPHI	*Descurainia*	*sophia*	Brassicaceae	E	1		1		I	E
ECHCRUSG	*Echinochloa*	*crus-galli*	Poaceae	M	1	1		1	I	E
ELACOMMU	*Elaeagnus*	*commutata*	Elaeagnaceae	E					N	N
ELEACICU	*Eleocharis*	*acicularis*	Cyperaceae	M	1	1	1	1	N	N
ELEPALUS	*Eleocharis*	*palustris*	Cyperaceae	M	1	1	1	1	N	N
ELYREPEN	*Elymus*	*repens*	Poaceae	M		1		1	I	E
ELYTRACH	*Elymus*	*trachycaulus*	Poaceae	M	1	1	1	1	N	N
EPICAMPE	*Epilobium*	*campestre*	Onagraceae	E			1		N	N
EPICILIA	*Epilobium*	*ciliatum*	Onagraceae	E	1	1	1	1	N	N
EPILEPTO	*Epilobium*	*leptophyllum*	Onagraceae	E	1		1		N	N
EPIPALUS	*Epilobium*	*palustre*	Onagraceae	E		1	1	1	N	N
EQUARVEN	*Equisetum*	*arvense*	Equisetaceae	SV		1	1	1	N	N
EQUFLUVI	*Equisetum*	*fluviatile*	Equisetaceae	SV				1	N	N
EQUHYMAL	*Equisetum*	*hyemale*	Equisetaceae	SV	1				N	N
EQUPALUS	*Equisetum*	*palustre*	Equisetaceae	SV					N	N
EQUPRATE	*Equisetum*	*pratense*	Equisetaceae	SV	1	1		1	N	N
ERIGRACI	*Eriophorum*	*gracile*	Cyperaceae	M					N	N
ERILONCH	*Erigeron*	*lonchophyllus*	Asteraceae	E	1				N	N
ERIPHILA	*Erigeron*	*philadelphicus*	Asteraceae	E		1			N	N
ERUGALLI	*Erucastrum*	*gallicum*	Brassicaceae	E			1		I	E
ERYCHEIR	*Erysimum*	*cheiranthoides*	Brassicaceae	E	1	1		1	N	N

		Taxonomy			Detection				Status	
Code	Genus	Specific Epithet	Family	Group	GL 2014	PL 2014	GL 2015	PL 2015	Native Status	Exotic ACIMS
EURCONSP	*Eurybia*	*conspicua*	Asteraceae	E				1	N	N
FAGESCUL	*Fagopyrum*	*esculentum*	Polygonaceae	E				1	I	E
FALCONVO	*Fallopia*	*convolvulus*	Polygonaceae	E	1	1	1		I	E
FALSCAND	*Fallopia*	*scandens*	Polygonaceae	E	1				I	E
FESSAXIM	*Festuca*	*saximontana*	Poaceae	M		1			N	N
FRAVESCA	*Fragaria*	*vesca*	Rosaceae	E		1		1	N	N
FRAVIRGI	*Fragaria*	*virginiana*	Rosaceae	E		1		1	N	N
GALTETRA	*Galeopsis*	*tetrahit*	Lamiaceae	E		1		1	I	E
GALTRIFI	*Galium*	*trifidum*	Rubiaceae	E		1	1	1	N	N
GALTRIFL	*Galium*	*triflorum*	Rubiaceae	E		1		1	N	N
GEUALEPP	*Geum*	*aleppicum*	Rosaceae	E	1	1		1	N	N
GEUMACRO	*Geum*	*macrophyllum*	Rosaceae	E		1		1	N	N
GEURIVAL	*Geum*	*rivale*	Rosaceae	E		1			N	N
GLYBOREA	*Glyceria*	*borealis*	Poaceae	M	1				N	N
GLYGRAND	*Glyceria*	*grandis*	Poaceae	M	1	1	1	1	N	N
GLYSTRIA	*Glyceria*	*striata*	Poaceae	M		1		1	N	N
GRANEGLE	*Gratiola*	*neglecta*	Plantaginaceae	E		1			N	N
GRISQUAR	*Grindelia*	*squarrosa*	Asteraceae	E	1		1		N	N
HIEUMBAL	*Hieracium*	*umbellatum*	Asteraceae	E				1	N	N
HIPVULGA	*Hippuris*	*vulgaris*	Plantaginaceae	E	1		1		N	N
HORJUBAT	*Hordeum*	*jubatum*	Poaceae	M	1	1	1	1	N	N
HORVULGA	*Hordeum*	*vulgare*	Poaceae	M			1	1	I	E
JUNBALTI	*Juncus*	*balticus*	Juncaceae	M	1	1	1	1	N	N
JUNLONGI	*Juncus*	*longistylus*	Juncaceae	M		1			N	N
JUNNODOS	*Juncus*	*nodosus*	Juncaceae	M		1		1	N	N
JUNVASEY	*Juncus*	*vaseyi*	Juncaceae	M		1			N	N
KRALANAT	*Krascheninnikovia*	*lanata*	Amaranthaceae	E	1		1		N	N
LACSERRI	*Lactuca*	*serriola*	Asteraceae	E	1		1		I	E

	Taxonomy				Detection				Status	
Code	Genus	Specific Epithet	Family	Group	GL 2014	PL 2014	GL 2015	PL 2015	Native Status	Exotic ACIMS
LATOCHRO	*Lathyrus*	*othroleucus*	Fabaceae	E		1			N	N
LEMMINOR	*Lemna*	*minor*	Araceae	M	1	1		1	N	N
LEMTRISU	*Lemna*	*trisulca*	Araceae	M					N	N
LINUSITA	*Linum*	*usitatissimum*	Linaceae	E	1				I	E
LYCASPER	*Lycopus*	*asper*	Lamiaceae	E		1		1	N	N
LYSCILIA	*Lysimachia*	*ciliata*	Primulaceae	E					N	N
LYSMARIT	*Lysimachia*	*maritima*	Primulaceae	E		1			N	N
LYSTHYRS	*Lysimachia*	*thyrsiflora*	Primulaceae	E	1			1	N	N
MAISTELL	*Maianthemum*	*stellatum*	Asparagaceae	M	1	1			N	N
MALNEGLE	*Malva*	*neglecta*	Malvaceae	E	1				I	E
MEDSATIV	*Medicago*	*sativa*	Fabaceae	E	1				I	E
MELALBUS	*Melilotus*	*albus*	Fabaceae	E		1	1		I	E
MENARVEN	*Mentha*	*arvensis*	Lamiaceae	E	1	1	1	1	N	N
MONNUTTA	*Monolepis*	*nuttalliana*	Amaranthaceae	E					N	N
MUHRICHA	*Muhlenbergia*	*richardsonis*	Poaceae	M			1		N	N
MULOBLON	*Mulgedium*	*oblongifolium*	Asteraceae	E					N	N
PENPROCE	*Penstemon*	*procerus*	Plantaginaceae	E			1		N	N
PERAMPHI	*Persicaria*	*amphibia*	Polygonaceae	E	1	1	1	1	N	N
PERLAPAT	*Persicaria*	*lapathifolia*	Polygonaceae	E	1	1	1	1	I	N
PETFRIGI	*Petasites*	*frigidus*	Asteraceae	E		1		1	N	N
PHAARUND	*Phalaris*	*arundinacea*	Poaceae	M	1	1	1	1	N	N
PHLPRATE	*Phleum*	*pratense*	Poaceae	M	1	1	1		I	E
PLAHYPER	*Platanthera*	*hyperborea*	Orchidaceae	M		1			N	N
PLAMAJOR	*Plantago*	*major*	Plantaginaceae	E		1	1	1	I	E
PLASCOUL	*Plagiobothrys*	*scouleri*	Boraginaceae	E	1	1	1	1	N	N
POAPALUS	*Poa*	*palustris*	Poaceae	M	1	1	1	1	N	N
POAPRATE	*Poa*	*pratensis*	Poaceae	M	1	1	1		N	N
POLAVICU	*Polygonum*	*aviculare*	Polygonaceae	E	1		1		I	E

		Taxonomy			Detection				Status	
Code	Genus	Specific Epithet	Family	Group	GL 2014	PL 2014	GL 2015	PL 2015	Native Status	Exotic ACIMS
POLRAMOS	*Polygonum*	*ramosissimum*	Polygonaceae	E	1				N	N
POPBALSA	*Populus*	*balsamifera*	Salicaceae	E					N	N
POPTREMU	*Populus*	*tremuloides*	Salicaceae	E		1		1	N	N
POTANSER	*Potentilla*	*anserina*	Rosaceae	E	1	1	1	1	N	N
POTGRAMI	*Potamogeton*	*gramineus*	Potamogetonaceae	M	1				N	N
POTNORVE	*Potentilla*	*norvegica*	Rosaceae	E	1	1	1	1	N	N
POTRICHA	*Potamogeton*	*richardsonii*	Potamogetonaceae	M	1		1		N	N
POTRIVAL	*Potentilla*	*rivalis*	Rosaceae	E			1		N	N
PYRASARI	*Pyrola*	*asarifolia*	Pyrolaceae	E					N	N
RANAQUAT	*Ranunculus*	*aquatilis*	Ranunculaceae	E	1	1			N	N
RANCYMBA	*Ranunculus*	*cymbalaria*	Ranunculaceae	E	1	1		1	N	N
RANGMELI	*Ranunculus*	*gmelinii*	Ranunculaceae	E	1	1	1	1	N	N
RANMACOU	*Ranunculus*	*macounii*	Ranunculaceae	E				1	N	N
RANSCELE	*Ranunculus*	*sceleratus*	Ranunculaceae	E	1	1		1	N	N
RIBLACUS	*Ribes*	*lacustre*	Grossulariaceae	E				1	N	N
RIBOXYAC	*Ribes*	*oxyacanthoides*	Grossulariaceae	E		1			N	N
RORPALUS	*Rorippa*	*palustris*	Brassicaceae	E	1	1	1	1	N	N
ROSACICU	*Rosa*	*acicularis*	Rosaceae	E	1	1	1	1	N	N
RUBPUBES	*Rubus*	*pubescens*	Rosaceae	E		1			N	N
RUBSACHA	*Rubus*	*sachalinensis*	Rosaceae	E		1			N	N
RUMBRITA	*Rumex*	*britannica*	Polygonaceae	E	1				N	N
RUMCRISP	*Rumex*	*crispus*	Polygonaceae	E	1	1	1	1	I	E
RUMFUEGI	*Rumex*	*fueginus*	Polygonaceae	E	1	1	1	1	N	N
RUMOCCID	*Rumex*	*occidentalis*	Polygonaceae	E	1	1		1	N	N
RUMSALIC	*Rumex*	*salicifolius*	Polygonaceae	E	1		1		N	N
SAGCUNEA	*Sagittaria*	*cuneata*	Alismataceae	M	1	1			N	N
SALBEBBI	*Salix*	*bebbiana*	Salicaceae	E					N	N
SALDISCO	*Salix*	*discolor*	Salicaceae	E		1			N	N

	Taxonomy				Detection				Status	
Code	Genus	Specific Epithet	Family	Group	GL 2014	PL 2014	GL 2015	PL 2015	Native Status	Exotic ACIMS
SALEXIGU	*Salix*	*exigua*	Salicaceae	E		1		1	N	N
SALLASIA	*Salix*	*lasiandra*	Salicaceae	E	1	1	1	1	N	N
SALLUCID	*Salix*	*lucida*	Salicaceae	E		1			N	N
SALMACCA	*Salix*	*maccalliana*	Salicaceae	E					N	N
SALPETIO	*Salix*	*petiolaris*	Salicaceae	E					N	N
SALPLANI	*Salix*	*planifolia*	Salicaceae	E		1		1	N	N
SALPSEUD	*Salix*	*pseudomonticola*	Salicaceae	E		1			N	N
SALPYRIF	*Salix*	*pyrifolia*	Salicaceae	E					N	N
SALRUBRA	*Salicornia*	*rubra*	Amaranthaceae	E	1				N	N
SALSERIS	*Salix*	*serissima*	Salicaceae	E		1		1	N	N
SCHACUTU	*Schoenoplectus*	*acutus*	Cyperaceae	M	1		1		N	N
SCHPUNGE	*Schoenoplectus*	*pungens*	Cyperaceae	M		1		1	N	N
SCHTABER	*Schoenoplectus*	*tabernaemontani*	Cyperaceae	M	1	1	1	1	N	N
SCIMICRO	*Scirpus*	*microcarpus*	Cyperaceae	M				1	N	N
SCOFESTU	*Scolochloa*	*festucacea*	Poaceae	M	1	1	1	1	N	N
SCUGALER	*Scutellaria*	*galericulata*	Lamiaceae	E	1	1		1	N	N
SENVULGA	*Senecio*	*vulgaris*	Asteraceae	E	1				I	E
SISMONTA	*Sisyrinchium*	*montanum*	Iridaceae	M					N	N
SIUSUAVE	*Sium*	*suave*	Apiaceae	E	1	1	1	1	N	N
SOLALTIS	*Solidago*	*altissima*	Asteraceae	E	1	1			N	N
SONARVEN	*Sonchus*	*arvensis*	Asteraceae	E	1	1	1	1	I	E
SONASPER	*Sonchus*	*asper*	Asteraceae	E	1	1	1		I	E
SONOLERA	*Sonchus*	*oleraceus*	Asteraceae	E	1		1		I	E
SPAANGUS	*Sparganium*	*angustifolium*	Typhaceae	M					N	N
SPAEURYC	*Sparganium*	*eurycarpum*	Typhaceae	M					N	N
SPESALIN	*Spergularia*	*salina*	Caryophyllaceae	E	1				N	N
SPHINTER	*Sphenopholis*	*intermedia*	Poaceae	M					N	N
SPOCRYPT	*Sporobolus*	*cryptandrus*	Poaceae	M					N	N

	Taxonomy				Detection				Status	
Code	Genus	Specific Epithet	Family	Group	GL 2014	PL 2014	GL 2015	PL 2015	Native Status	Exotic ACIMS
STAPILOS	*Stachys*	*pilosa*	Lamiaceae	E	1	1	1	1	N	N
STELONGI	*Stellaria*	*longifolia*	Caryophyllaceae	E		1			N	N
STEMEDIA	*Stellaria*	*media*	Caryophyllaceae	E					I	E
SUACALCE	*Suaeda*	*calceoliformis*	Amaranthaceae	E	1				N	N
SYMBOREA	*Symphyotrichum*	*boreale*	Asteraceae	E	1	1			N	N
SYMERICO	*Symphyotrichum*	*ericoides*	Asteraceae	E	1				N	N
SYMLANCE	*Symphyotrichum*	*lanceolatum*	Asteraceae	E	1	1	1	1	N	N
SYMOCCID	*Symphoricarpos*	*occidentalis*	Caprifoliaceae	E	1		1	1	N	N
SYMPUNIC	*Symphyotrichum*	*puniceum*	Asteraceae	E		1			N	N
TANVULGA	*Tanacetum*	*vulgare*	Asteraceae	E					I	E
TAROFFIC	*Taraxacum*	*officinale*	Asteraceae	E	1	1	1	1	I	E
TEPPALUS	*Tephroseris*	*palustris*	Asteraceae	E				1	N	N
THLARVEN	*Thlaspi*	*arvense*	Brassicaceae	E	1	1	1	1	I	E
TRADUBIU	*Tragopogon*	*dubius*	Asteraceae	E			1		I	E
TRIHYBRI	*Trifolium*	*hybridum*	Fabaceae	E		1		1	I	E
TRIMARIT	*Triglochin*	*maritima*	Juncaginaceae	M	1	1			N	N
TYPLATIF	*Typha*	*latifolia*	Typhaceae	M	1	1	1	1	N	N
URTDIOCA	*Urtica*	*dioica*	Urticaceae	E		1		1	N	N
UTRVULGA	*Utricularia*	*vulgaris*	Lentibulariaceae	E	1				N	N
VERPEREG	*Veronica*	*peregrina*	Plantaginaceae	E		1			N	N
VERSCUTE	*Veronica*	*scutellata*	Plantaginaceae	E	1	1	1	1	N	N
VICAMERI	*Vicia*	*americana*	Fabaceae	E		1		1	N	N
VIOADUNC	*Viola*	*adunca*	Violaceae	E		1			N	N
VIOCANAD	*Viola*	*canadensis*	Violaceae	E					N	N
VIOSOROR	*Viola*	*sororia*	Violaceae	E				1	N	N

Table C: Plants species and their endemism and conservation traits

Code	S Rank	G Rank	Watched	Rare	Weed	Noxious	Grassland Region CC	Parkland Region CC	Sensitivity/Tolerance (GL)	Sensitivity/Tolerance (PL)
ACHALPIN	S5	G5?	-	-	-	-	ND	3		
ACHMILLE	S5	G5	-	-	-	-	5	5		
ACOCALAM	S3	G5	-	-	-	-	ND	8		S
AGRCRIST	SNA	GNR	-	-	-	-	0	ND	T	
AGRGIGAN	SNA	GNR	-	-	-	-	ND	0		T
AGRSCABR	S5	G5	-	-	-	-	2	2	T	T
AGRSTRIA	S4	G5	-	-	-	-	ND	5		
ALITRIVI	S5?	G5	-	-	-	-	6	4	S	
ALOAEQUA	S5	G5	-	-	-	-	4	4		
ALOPRATE	SNA	GNR	-	-	-	-	ND	0		T
AMARETRO	SNA	GNR	-	-	W	-	0	ND	T	
AMEALNIF	S5	G5	-	-	-	-	ND	3		
ANAMINIM	S2S3	G5	T	R	-	-	6	ND	S	
ANECANAD	S5	G5	-	-	-	-	4	ND		
ANTPARVI	S5	G5	-	-	-	-	ND	4		
ARNCHAMI	S5	G5	-	-	-	-	5	ND		
ARTBIENN	S5	G5	-	-	-	-	2	2	T	T
ARTCAMPE	S5	G5	-	-	-	-	2	ND	T	
ARTLONGI	S3	G5	-	-	-	-	7	7	S	S
ARTLUDOV	S5	G5	-	-	-	-	3	3		
ATRPROST	SNA	G5	-	-	-	-	0	ND	T	
AVEFATUA	SNA	GNR	-	-	W	-	0	ND	T	
BECSYZIG	S5	G5	-	-	-	-	4	2		T
BIDCERNU	S5	G5	-	-	-	-	4	4		
BOLMARIT	S4	G5	-	-	-	-	ND	6		S
BRANAPUS	SNA	GNR	-	-	W	-	0	ND	T	
BROINERM	SNA	GNR	-	-	-	-	0	0	T	T
CALCANAD	S5	G5	-	-	-	-	5	3		

Code	S Rank	G Rank	Watched	Rare	Weed	Noxious	Grassland Region CC	Parkland Region CC	Sensitivity/Tolerance (GL)	Sensitivity/Tolerance (PL)
CALIPALU	S5	G5	-	-	-	-	6	4	S	
CALLPALU	S4S5	G5	-	-	-	-	ND	7		S
CALSTRIC	S5	G5T5	-	-	-	-	6	4	S	
CALTPALU	S5	G5	-	-	-	-	ND	6		S
CAPBURSA	SNA	GNR	-	-	W	-	0	0	T	T
CARAQUAT	S5	G5	-	-	-	-	5	4		
CARATHER	S5	G5	-	-	-	-	5	5		
CARATHRO	S4	G5	-	-	-	-	4	4		
CARBEBBI	S5	G5	-	-	-	-	7	3	S	
CARBREVI	S3	G5?	-	-	-	-	4	ND		
CARCARVI	SNA	GNR	-	-	W	-	ND	0		T
CARDIAND	S5	G5	-	-	-	-	ND	5		
CARLACUS	S4	G5	-	-	-	-	ND	7		S
CARPELLI	S5	G5	-	-	-	-	4	5		
CARPRAEG	S5	G5	-	-	-	-	4	4		
CARPRATI	S5	G5	-	-	-	-	ND	6		S
CARRETRO	S4	G5	-	R	-	-	7	ND	S	
CARSARTW	S4	G4G5	-	-	-	-	ND	5		
CARSYCHN	S5?	G4	-	-	-	-	ND	5		
CARUTRIC	S5	G5	-	-	-	-	3	5		
CERARVEN	S5	G5	-	-	-	-	ND	3		
CHAANGUS	S5	G5	-	-	-	-	ND	1		T
CHEALBUM	SNA	G5	-	-	W	-	0	0	T	T
CHECAPIT	S5	G5	-	-	-	-	1	1	T	T
CHERUBRU	S4	G5	-	-	-	-	ND	4		
CICMACUL	S5	G5	-	-	W	-	ND	4		
CIRARVEN	SNA	GNR	-	-	W	N	0	0	T	T
CIRVULGA	SNA	GNR	-	-	W	-	0	ND	T	
COLLINEA	S5	G5	-	-	-	-	3	ND		

Code	S Rank	G Rank	Watched	Rare	Weed	Noxious	Grassland Region CC	Parkland Region CC	Sensitivity/Tolerance (GL)	Sensitivity/Tolerance (PL)
COMPALUS	S5	G5	-	-	-	-	ND	7		S
CORSERIC	S5	G5	-	-	-	-	ND	3		
CRETECTO	SNA	GNR	-	-	W	-	0	0	T	T
DESCESPI	S5	G5	-	-	-	-	7	4	S	
DESSOPHI	SNA	GNR	-	-	W	-	0	0	T	T
ECHCRUSG	SNA	GNR	-	-	W	-	0	0	T	T
ELACOMMU	S5	G5	-	-	-	-	ND	5		
ELEACICU	S5	G5	-	-	-	-	4	4		
ELEPALUS	S5	G5	-	-	-	-	4	4		
ELYREPEN	SNA	GNR	-	-	-	-	ND	0		T
ELYTRACH	S5	G5	-	-	-	-	5	2		T
EPICAMPE	S3	G5	T	R	-	-	10	ND	S	
EPICILIA	S5	G5	-	-	-	-	3	2		T
EPILEPTO	S3	G5	-	-	-	-	6	ND	S	
EPIPALUS	S4	G5	-	-	-	-	7	3	S	
EQUARVEN	S5	G5	-	-	W	-	2	1	T	T
EQUFLUVI	S5	G5	-	-	-	-	ND	5		
EQUHYMAL	S5	G5T5	-	-	-	-	3	ND		
EQUPALUS	S5	G5	-	-	-	-	ND	5		
EQUPRATE	S5	G5	-	-	-	-	6	5	S	
ERIGRACI	S4	G5	-	-	-	-	ND	7		S
ERILONCH	S5	G5	-	-	-	-	4	ND		
ERIPHILA	S5	G5	-	-	-	-	ND	2		T
ERUGALLI	SNA	GNR	-	-	-	-	0	ND	T	
ERYCHEIR	S5	G5	-	-	-	-	0	0	T	T
EURCONSP	S5	G5	-	-	-	-	ND	4		
FAGESCUL	SNA	GNR	-	-	-	-	0	0	T	T
FALCONVO	SNA	GNR	-	-	W	-	0	0	T	T
FALSCAND	SNA	GNR	-	-	-	-	0	0	T	T

Code	S Rank	G Rank	Watched	Rare	Weed	Noxious	Grassland Region CC	Parkland Region CC	Sensitivity/Tolerance (GL)	Sensitivity/Tolerance (PL)
FESSAXIM	S5	G5	-	-	-	-	ND	8		S
FRAVESCA	S4	G5	-	-	-	-	ND	4		
FRAVIRGI	S5	G5	-	-	-	-	ND	1		T
GALTETRA	SNA	GNR	-	-	W	-	ND	0		T
GALTRIFI	S5	G5	-	-	-	-	6	4	S	
GALTRIFL	S5	G5	-	-	-	-	ND	3		
GEUALEPP	S5	G5	-	-	-	-	6	3	S	
GEUMACRO	S5	G5	-	-	-	-	ND	6		S
GEURIVAL	S5	G5	-	-	-	-	ND	6		S
GLYBOREA	S4	G5	-	-	-	-	6	ND	S	
GLYGRAND	S5	G5	-	-	-	-	7	4	S	
GLYSTRIA	S5?	G5	-	-	-	-	ND	4		
GRANEGLE	S3	G5	T	R	-	-	ND	4		
GRISQUAR	S4S5	G5	-	-	W	-	2	ND	T	
HIEUMBAL	S5	G5	-	-	-	-	ND	2		T
HIPVULGA	S5	G5	-	-	-	-	6	ND	S	
HORJUBAT	S5	G5	-	-	W	-	2	1	T	T
HORVULGA	SNA	GNR	-	-	W	-	0	0	T	T
JUNBALTI	S5	G5	-	-	-	-	3	3		
JUNLONGI	S4	G5	-	-	-	-	ND	5		
JUNNODOS	S5	G5	-	-	-	-	ND	4		
JUNVASEY	S4	G5?	-	-	-	-	ND	5		
KRALANAT	S5	G5	-	-	-	-	8	ND	S	
LACSERRI	SNA	GNR	-	-	W	-	0	ND	T	
LATOCHRO	S5	G5	-	-	-	-	ND	8		S
LEMMINOR	missing	missing	-	-	-	-	2	4	T	
LEMTRISU	S5?	G5	-	-	-	-	ND	4		
LINUSITA	SNA	GNR	-	-	W	-	0	ND	T	
LYCASPER	S3	G5	-	-	-	-	ND	4		

Code	S Rank	G Rank	Watched	Rare	Weed	Noxious	Grassland Region CC	Parkland Region CC	Sensitivity/Tolerance (GL)	Sensitivity/Tolerance (PL)
LYSCILIA	S4	G5	-	-	-	-	ND	7		S
LYSMARIT	S4	G5	-	-	-	-	ND	6		S
LYSTHYRS	S4	G5	-	-	-	-	8	6	S	S
MAISTELL	S5	G5	-	-	-	-	4	5		
MALNEGLE	SNA	GNR	-	-	W	-	0	ND	T	
MEDSATIV	SNA	GNR	-	-	-	-	0	0	T	T
MELALBUS	SNA	G5	-	-	-	-	0	0	T	T
MENARVEN	S5	G5	-	-	-	-	3	4		
MONNUTTA	S5	G5	-	-	-	-	ND	1		T
MUHRICHA	S5	G5	-	-	-	-	4	ND		
MULOBLON	S5	G5T5	-	-	-	-	ND	4		
PENPROCE	S5	G5	-	-	-	-	5	ND		
PERAMPHI	S5	G5	-	-	W	-	6	2	S	T
PERLAPAT	S5	G5	-	-	W	-	2	2	T	T
PETFRIGI	S5	G5	-	-	-	-	ND	2		T
PHAARUND	S5	G5	-	-	-	-	0	0	T	T
PHLPRATE	SNA	GNR	-	-	-	-	0	0	T	T
PLAHYPER	S5	G5	-	-	-	-	ND	5		
PLAMAJOR	SNA	GNR	-	-	W	-	0	0	T	T
PLASCOUL	S3	G5	-	-	-	-	2	2	T	T
POAPALUS	S5	G5	-	-	-	-	3	3		
POAPRATE	S5	G5	-	-	-	-	0	0	T	T
POLAVICU	SNA	GNR	-	-	W	-	0	0	T	T
POLRAMOS	S3	G5	-	-	-	-	3	ND		
POPBALSA	S5	G5	-	-	-	-	ND	5		
POPTREMU	S5	G5	-	-	-	-	ND	5		
POTANSER	S5	G5	-	-	-	-	3	3		
POTGRAMI	S4	G5	-	-	-	-	6	ND	S	
POTNORVE	S5	G5	-	-	W	-	2	2	T	T

Code	S Rank	G Rank	Watched	Rare	Weed	Noxious	Grassland Region CC	Parkland Region CC	Sensitivity/Tolerance (GL)	Sensitivity/Tolerance (PL)
POTRICHA	S5	G5	-	-	-	-	6	ND	S	
POTRIVAL	S4	G5	-	-	-	-	4	4		
PYRASARI	S5	G5	-	-	-	-	ND	6		S
RANAQUAT	S5	G5	-	-	-	-	4	5		
RANCYMBA	S5	G5	-	-	-	-	3	4		
RANGMELI	S5	G5	-	-	-	-	4	4		
RANMACOU	S5	G5	-	-	-	-	ND	5		
RANSCELE	S5	G5	-	-	-	-	4	3		
RIBLACUS	S5	G5	-	-	-	-	ND	6		S
RIBOXYAC	S5	G5	-	-	-	-	ND	3		
RORPALUS	S5	G5	-	-	-	-	4	4		
ROSACICU	S5	G5	-	-	W	-	3	3		
RUBPUBES	S5	G5	-	-	-	-	ND	5		
RUBSACHA	S5	G5	-	-	-	-	ND	1		T
RUMBRITA	S3	G5	-	-	-	-	4	ND		
RUMCRISP	SNA	GNR	-	-	-	-	0	0	T	T
RUMFUEGI	S5	G5	-	-	-	-	6	2	S	T
RUMOCCID	S5	G5T5	-	-	-	-	4	4		
RUMSALIC	S5	G5	-	-	-	-	7	ND	S	
SAGCUNEA	S5	G5	-	-	-	-	7	5	S	
SALBEBBI	S5	G5	-	-	-	-	ND	2		T
SALDISCO	S5	G5	-	-	-	-	ND	2		T
SALEXIGU	S3S4	G5	-	-	-	-	ND	2		T
SALLASIA	S5	G5T5	-	-	-	-	4	5		
SALLUCID	S3	G5T5	-	-	-	-	ND	5		
SALMACCA	S4	G5?	-	-	-	-	ND	3		
SALPETIO	S5	G5	-	-	-	-	ND	4		
SALPLANI	S5	G5	-	-	-	-	ND	4		
SALPSEUD	S4	G4G5	-	-	-	-	ND	4		

Code	S Rank	G Rank	Watched	Rare	Weed	Noxious	Grassland Region CC	Parkland Region CC	Sensitivity/Tolerance (GL)	Sensitivity/Tolerance (PL)
SALPYRIF	S5	S5	-	-	-	-	ND	6		S
SALRUBRA	S5	G5	-	-	-	-	7	ND	S	
SALSERIS	S4	G4	-	-	-	-	ND	6		S
SCHACUTU	S5?	G5	-	-	-	-	5	ND		
SCHPUNGE	S4	G5	-	-	-	-	ND	6		S
SCHTABER	S5	G5	-	-	-	-	6	4	S	
SCIMICRO	S5	G5	-	-	-	-	ND	3		
SCOFESTU	S4	G5	-	-	-	-	4	4		
SCUGALER	S5	G5	-	-	-	-	6	6	S	S
SENVULGA	SNA	GNR	-	-	W	-	0	ND	T	
SISMONTA	S5	G5	-	-	-	-	ND	5		
SIUSUAVE	S5	G5	-	-	-	-	7	5	S	
SOLALTIS	S5	GNR	-	-	-	-	3	2		T
SONARVEN	SNA	GNR	-	-	W	N	0	0	T	T
SONASPER	SNA	GNR	-	-	W	-	0	0	T	T
SONOLERA	SNA	GNR	-	-	-	-	0	ND	T	
SPAANGUS	S4	G5	-	-	-	-	ND	6		S
SPAEURYC	S4	G5	-	-	-	-	ND	5		
SPESALIN	S3	G5	-	R	-	-	8	ND	S	
SPHINTER	S4	G5	-	-	-	-	ND	7		S
SPOCRYPT	S3	G5	-	-	-	-	ND	6		S
STAPILOS	S5	G5	-	-	-	-	6	4	S	
STELONGI	S5	G5	-	-	-	-	ND	4		
STEMEDIA	SNA	GNR	-	-	W	-	ND	0		T
SUACALCE	S5	G5	-	-	-	-	3	ND		
SYMBOREA	S5	G5	-	-	-	-	7	6	S	S
SYMERICO	S5	G5T5	-	-	-	-	6	5	S	
SYMLANCE	S5	G5T5	-	-	-	-	4	6		S
SYMOCCID	S5	G5	-	-	-	-	4	4		

Code	S Rank	G Rank	Watched	Rare	Weed	Noxious	Grassland Region CC	Parkland Region CC	Sensitivity/Tolerance (GL)	Sensitivity/Tolerance (PL)
SYMPUNIC	S4	G5	-	-	-	-	ND	5		
TANVULGA	SNA	GNR	-	-	W	N	ND	0		T
TAROFFIC	SNA	GNR	-	-	W	-	0	0	T	T
TEPPALUS	S5	G5	-	-	-	-	ND	3		
THLARVEN	SNA	GNR	-	-	W	-	0	0	T	T
TRADUBIU	SNA	GNR	-	-	-	-	0	ND	T	
TRIHYBRI	SNA	G5	-	-	-	-	ND	0		T
TRIMARIT	S5	G5	-	-	-	-	8	5	S	
TYPLATIF	S5	G5	-	-	-	-	3	2		T
URTDIOCA	S5	G5	-	-	W	-	ND	3		
UTRVULGA	S5	G5	-	-	-	-	5	ND		
VERPEREG	S5	G5	-	-	-	-	ND	3		
VERSCUTE	S3	G5	-	-	-	-	6	3	S	
VICAMERI	S5	G5	-	-	-	-	ND	2		T
VIOADUNC	S5	G5	-	-	-	-	ND	5		
VIOCANAD	S5	G5	-	-	-	-	ND	4		
VIOSOROR	S4	G5	-	-	-	-	ND	7		S

Table D: Plant species and their autecology

Code	Lifecycle	Wetland Status	Habit	Form	Vegetative Reproduction	Nitrogen-Fixing	Recalcitrant Litter	Dispersal Mechanism
ACHALPIN	P	UPL	F	GC	R			A
ACHMILLE	P	FACU	F	GC	R			A
ACOCALAM	P	OBL	G	NE	R			AHZ
AGRCRIST	P	UPL	G	NE	-			A
AGRGIGAN	P	FACW	G	NE	R, S			A
AGRSCABR	P	FAC	G	NE	-			A
AGRSTRIA	P	FACU	F	GC	R			Z
ALITRIVI	P	OBL	F	BE	-			H
ALOAEQUA	P	OBL	G	NE	-			AZ
ALOPRATE	P	FACW	G	NE	-			AZ
AMARETRO	A	FACU	F	GC	-			AH
AMEALNIF	P	FACU	S	TS	S			Z
ANAMINIM	A	UPL	F	GC	-			H
ANECANAD	P	FACW	F	GC	-			AZ
ANTPARVI	P	UPL	F	GC	S			A
ARNCHAMI	P	FACW	F	GC	R			A
ARTBIENN	B	FACU	F	GC	-			AHZ
ARTCAMPE	B	UPL	F	GC	-			A
ARTLONGI	P	UPL	F	GC	-			A
ARTLUDOV	P	UPL	F	GC	R			A
ATRPROST	A	FACW	F	GC	-			AHZ
AVEFATUA	A	UPL	G	NE	-			Z
BECSYZIG	A	OBL	G	NE	-			HZ
BIDCERNU	A	OBL	F	BE	-			Z
BOLMARIT	P	OBL	G	NE	R		1	H
BRANAPUS	B	UPL	F	GC	-			AZ
BROINERM	P	UPL	G	NE	R			A
CALCANAD	P	FACW	G	NE	R			A

Code	Lifecycle	Wetland Status	Habit	Form	Vegetative Reproduction	Nitrogen-Fixing	Recalcitrant Litter	Dispersal Mechanism
CALIPALU	P	OBL	A	F	-			H
CALLPALU	P	OBL	F	BE	R			H
CALSTRIC	P	FACW	G	NE	R			AZ
CALTPALU	P	OBL	F	BE	-			H
CAPBURSA	A	FACU	F	GC	-			Z
CARAQUAT	P	OBL	G	NE	R			HZ
CARATHER	P	OBL	G	NE	R			HZ
CARATHRO	P	FACW	G	NE	-			H
CARBEBBI	P	OBL	G	NE	-			H
CARBREVI	P	FAC	G	NE	-			H
CARCARVI	B	UPL	F	GC	-			H
CARDIAND	P	OBL	G	NE	-			HZ
CARLACUS	P	OBL	G	NE	R			H
CARPELLI	P	OBL	G	NE	R			H
CARPRAEG	P	FACW	G	NE	R			H
CARPRATI	P	FAC	G	NE	R			H
CARRETRO	P	OBL	G	NE	-			H
CARSARTW	P	FACW	G	NE	R			H
CARSYCHN	P	FACW	G	NE	-			H
CARUTRIC	P	OBL	G	NE	R			H
CERARVEN	P	FACU	F	GC	-			A
CHAANGUS	P	FAC	F	GC	R			A
CHEALBUM	A	FACU	F	GC	-			HZ
CHECAPIT	A	UPL	F	GC	-			HZ
CHERUBRU	A	OBL	F	GC	-			HZ
CICMACUL	B	OBL	F	BE	-			AHZ
CIRARVEN	P	FACU	F	GC	R			AZ
CIRVULGA	B	UPL	F	GC	-			AZ
COLLINEA	A	FACU	F	GC	-			HZ

Code	Lifecycle	Wetland Status	Habit	Form	Vegetative Reproduction	Nitrogen-Fixing	Recalcitrant Litter	Dispersal Mechanism
COMPALUS	P	OBL	F	BE	R			Z
CORSERIC	P	UPL	S	TS	-			Z
CRETECTO	A	UPL	F	GC	-			AZ
DESCESPI	P	FACW	G	NE	-			A
DESSOPHI	B	UPL	F	GC	-			AHZ
ECHCRUSG	A	FAC	G	NE	-			H
ELACOMMU	P	UPL	S	TS	S	1		Z
ELEACICU	P	OBL	G	NE	R			AHZ
ELEPALUS	P	OBL	G	NE	R			AHZ
ELYREPEN	P	FACU	G	NE	R			Z
ELYTRACH	P	FACU	G	NE	-			Z
EPICAMPE	A	FACW	F	GC	-			Z
EPICILIA	P	FACW	F	GC	T			A
EPILEPTO	P	OBL	F	GC	S, T			A
EPIPALUS	P	OBL	F	GC	S, T			A
EQUARVEN	P	FAC	G	NE	R			AH
EQUFLUVI	P	OBL	G	NE	R			AH
EQUHYMAL	P	FACW	G	NE	R			AH
EQUPALUS	P	FACW	G	NE	R			AH
EQUPRATE	P	FACW	G	NE	R			AH
ERIGRACI	P	OBL	G	NE	-			A
ERILONCH	B	FACW	F	GC	-			A
ERIPHILA	B	FAC	F	GC	-			A
ERUGALLI	A	UPL	F	GC	-			AZ
ERYCHEIR	A	FACU	F	GC	-			AH
EURCONSP	P	UPL	F	GC	-			A
FAGESCUL	A	UPL	F	GC	-			A
FALCONVO	A	FACU	V	GC	-			AHZ
FALSCAND	P	FACU	V	GC	-			AHZ

Code	Lifecycle	Wetland Status	Habit	Form	Vegetative Reproduction	Nitrogen-Fixing	Recalcitrant Litter	Dispersal Mechanism
FESSAXIM	P	UPL	G	NE	-			AZ
FRAVESCA	P	UPL	F	GC	S			Z
FRAVIRGI	P	FACU	F	GC	S			Z
GALTETRA	A	FACU	F	GC	-			AH
GALTRIFI	P	OBL	F	GC	-			Z
GALTRIFL	P	FACU	F	GC	R			Z
GEUALEPP	P	FACU	F	GC	-			Z
GEUMACRO	P	FACW	F	GC	-			Z
GEURIVAL	P	FACW	F	GC	-			Z
GLYBOREA	P	OBL	G	NE	R			H
GLYGRAND	P	OBL	G	NE	R			H
GLYSTRIA	P	OBL	G	NE	R			H
GRANEGLE	A	OBL	F	GC	-			A
GRISQUAR	B	UPL	F	GC	-			A
HIEUMBAL	P	UPL	F	GC	-			AZ
HIPVULGA	P	OBL	A	BE	R			H
HORJUBAT	P	FACW	G	NE	-			AZ
HORVULGA	A	UPL	G	NE	-			Z
JUNBALTI	P	FACW	G	NE	R			A
JUNLONGI	P	FACW	G	NE	R			A
JUNNODOS	P	OBL	G	NE	R			A
JUNVASEY	P	FACW	G	NE	R			A
KRALANAT	P	UPL	F	GC	-			A
LACSERRI	A	FAC	F	GC	-			AH
LATOCHRO	P	UPL	V	GC	R	1		HZ
LEMMINOR	A	OBL	A	FF	-			H
LEMTRISU	A	OBL	A	FF	-			H
LINUSITA	A	UPL	F	GC	-			H
LYCASPER	P	OBL	F	GC	-			H

Code	Lifecycle	Wetland Status	Habit	Form	Vegetative Reproduction	Nitrogen-Fixing	Recalcitrant Litter	Dispersal Mechanism
LYSCILIA	P	FACW	F	GC	R			H
LYSMARIT	P	OBL	F	GC	R			HZ
LYSTHYRS	P	OBL	F	GC	R			H
MAISTELL	P	FACU	F	GC	R			Z
MALNEGLE	A	UPL	F	GC	-			H
MEDSATIV	P	FACU	F	GC	-	1		Z
MELALBUS	B	UPL	F	GC	-	1		HZ
MENARVEN	P	FACW	F	GC	R			H
MONNUTTA	A	FAC	F	GC	-			H
MUHRICHA	P	FAC	G	NE	R			A
MULOBLON	P	UPL	F	GC	-			A
PENPROCE	P	UPL	F	GC	-			A
PERAMPHI	P	OBL	F	BE	R		1	HZ
PERLAPAT	A	OBL	F	GC	-		1	HZ
PETFRIGI	P	FAC	F	GC	R			A
PHAARUND	P	FACW	G	NE	R		1	AH
PHLPRATE	P	FACU	G	NE	-			AZ
PLAHYPER	P	UPL	F	GC	-			AH
PLAMAJOR	P	FAC	F	GC	-			HZ
PLASCOUL	A	FACW	F	GC	-			Z
POAPALUS	P	FACW	G	NE	-			A
POAPRATE	P	FACU	G	NE	R			A
POLAVICU	A	FACU	F	GC	-			HZ
POLRAMOS	A	FACW	F	GC	-			HZ
POPBALSA	P	FACW	T	H	-			A
POPTREMU	P	FAC	T	H	R			A
POTANSER	P	FACW	F	GC	S			HZ
POTGRAMI	P	OBL	A	F	-			H
POTNORVE	A	FAC	F	GC	-			AZ

Code	Lifecycle	Wetland Status	Habit	Form	Vegetative Reproduction	Nitrogen-Fixing	Recalcitrant Litter	Dispersal Mechanism
POTRICHA	P	OBL	A	F	R			H
POTRIVAL	A	FACW	F	GC	R			AZ
PYRASARI	P	FACU	F	GC	R			H
RANAQUAT	P	OBL	A	BE	-			HZ
RANCYMBA	P	UPL	F	GC	-			HZ
RANGMELI	P	FACW	F	BE	S			HZ
RANMACOU	P	OBL	F	GC	-			HZ
RANSCELE	A	OBL	F	GC	-			HZ
RIBLACUS	P	FACW	S	LS	-			Z
RIBOXYAC	P	FACU	S	LS	-			Z
RORPALUS	A	OBL	F	GC	-			HZ
ROSACICU	P	FACU	S	LS	-			Z
RUBPUBES	P	FACW	F	GC	S			Z
RUBSACHA	P	FACU	S	TS	-			Z
RUMBRITA	P	OBL	F	GC	-			AHZ
RUMCRISP	P	FAC	F	GC	-			AHZ
RUMFUEGI	A	FACW	F	GC	-			HZ
RUMOCCID	P	OBL	F	GC	-			HZ
RUMSALIC	P	FACW	F	GC	-			HZ
SAGCUNEA	P	OBL	F	NE	-			HZ
SALBEBBI	P	FACW	S	TS	-			AH
SALDISCO	P	FACW	S	TS	-			AH
SALEXIGU	P	FACW	S	TS	-			AH
SALLASIA	P	FACW	S	TS	-			AH
SALLUCID	P	FACW	S	TS	-			AH
SALMACCA	P	OBL	S	TS	-			AH
SALPETIO	P	OBL	S	TS	-			AH
SALPLANI	P	OBL	S	TS	-			AH
SALPSEUD	P	FACW	S	TS	-			AH

Code	Lifecycle	Wetland Status	Habit	Form	Vegetative Reproduction	Nitrogen-Fixing	Recalcitrant Litter	Dispersal Mechanism
SALPYRIF	P	OBL	S	TS	-			AH
SALRUBRA	A	OBL	F	GC	-			HZ
SALSERIS	P	OBL	S	TS	-			AH
SCHACUTU	P	OBL	G	RE	R		1	AH
SCHPUNGE	P	OBL	G	NE	R		1	AH
SCHTABER	P	OBL	G	RE	R		1	AH
SCIMICRO	P	OBL	G	NE	R		1	H
SCOFESTU	P	OBL	G	NE	R			H
SCUGALER	P	OBL	F	GC	R			H
SENVULGA	A	FACU	F	GC	-			AZ
SISMONTA	P	FAC	F	GC	-			Z
SIUSUAVE	P	OBL	F	BE	-			H
SOLALTIS	P	FACU	F	GC	R			A
SONARVEN	P	FAC	F	GC	R			AZ
SONASPER	A	FAC	F	GC	-			AZ
SONOLERA	A	UPL	F	GC	-			AZ
SPAANGUS	P	OBL	G	NE	R		1	H
SPAEURYC	P	OBL	G	NE	R		1	H
SPESALIN	A	OBL	F	GC	-			AH
SPHINTER	P	FAC	G	NE	-			A
SPOCRYPT	P	FACU	G	NE	-			A
STAPILOS	P	FACW	F	GC	R			H
STELONGI	P	FACW	F	GC	-			HZ
STEMEDIA	A	FACU	F	GC	-			HZ
SUACALCE	A	FACW	F	GC	-			H
SYMBOREA	P	OBL	F	GC	R			A
SYMERICO	P	FACU	F	GC	-			A
SYMLANCE	P	FACW	F	GC	R			A
SYMOCCID	P	UPL	S	LS	-			Z

Code	Lifecycle	Wetland Status	Habit	Form	Vegetative Reproduction	Nitrogen-Fixing	Recalcitrant Litter	Dispersal Mechanism
SYMPUNIC	P	OBL	F	GC	-			A
TANVULGA	P	FACU	F	GC	R			AHZ
TAROFFIC	P	FACU	F	GC	-			AZ
TEPPALUS	A	UPL	F	GC	-			AZ
THLARVEN	A	FACU	F	GC	-			AHZ
TRADUBIU	A	UPL	F	GC	-			AZ
TRIHYBRI	P	FACU	F	GC	-	1		AZ
TRIMARIT	P	OBL	F	GC	R			H
TYPLATIF	P	OBL	G	RE	R		1	A
URTDIOCA	P	FAC	F	GC	R			AHZ
UTRVULGA	P	OBL	A	SU	-			H
VERPEREG	A	FACW	F	GC	-			AZ
VERSCUTE	P	OBL	F	GC	S			AZ
VICAMERI	P	FACU	V	GC	-	1		AZ
VIOADUNC	P	FACU	F	GC	R			AZ
VIOCANAD	P	FACU	F	GC	R			AZ
VIOSOROR	P	FAC	F	GC	R			AZ

Appendix 2.5 All potential metrics with their Spearman Rho and p-values

Includes metrics based on plant traits (Appendix 2.4) and plant families (Appendix 2.4). Note that nearly all metrics were calculated in 3 ways: as percent cover, presence or absence, or as the proportion of total richness.

Metric	Metric Variations	Spearman Rho	p-value
Achillea millefolium	Percent Cover	-0.12	0.853
	Presence/Absence	-0.14	0.771
Any species currently tracked by the ACIMS	Percent Cover	0.26	0.063
	Presence/Absence	0.26	0.062
	Proportion of Richness	0.26	0.063
Agropyron cristatum	Percent Cover	0.10	0.545
	Presence/Absence	0.10	0.545
Agrostis gigantea	Percent Cover	-0.16	0.283
	Presence/Absence	-0.16	0.283
Agrostis scabra	Percent Cover	0.27	0.094
	Presence/Absence	0.24	0.143
Agrimonia striata	Percent Cover	-0.07	0.691
	Presence/Absence	-0.06	0.722
Alismataceae family	Percent Cover	0.22	0.109
	Presence/Absence	0.21	0.127
	Proportion of Richness	0.22	0.109
Alisma triviale	Percent Cover	0.22	0.109
	Presence/Absence	0.21	0.127
Alopecurus aequalis	Percent Cover	0.29	0.055
	Presence/Absence	0.23	0.103
Amaranthaceae family	Percent Cover	0.05	0.826
	Presence/Absence	0.05	0.802
	Proportion of Richness	0.05	0.798
Amaranthus retroflexus	Percent Cover	0.24	0.105
	Presence/Absence	0.24	0.105
Anemone canadensis	Percent Cover	-0.07	0.618
	Presence/Absence	-0.07	0.618
Plant species whose primary means of propagule dispersal is Anemochory	Percent Cover	-0.03	0.879
	Presence/Absence	0.08	0.428
	Proportion of Richness	0.39	0.006
Plant species that are annual	Percent Cover	0.04	0.753
	Presence/Absence	0.23	0.143
	Proportion of Richness	0.40	0.009
Plant species that are annual or biennial	Percent Cover	0.05	0.730
	Presence/Absence	0.26	0.071
	Proportion of Richness	0.38	0.013

Metric	Metric Variations	Spearman Rho	p-value
Apiaceae family	Percent Cover	-0.16	0.567
	Presence/Absence	-0.15	0.698
	Proportion of Richness	-0.15	0.646
Plant species considered to be aquatic plants according to the Ontario Wetland Evaluation System (OWES)	Percent Cover	-0.27	0.024
	Presence/Absence	-0.25	0.029
	Proportion of Richness	-0.25	0.038
Araceae family	Percent Cover	-0.27	0.050
	Presence/Absence	-0.26	0.061
	Proportion of Richness	-0.24	0.082
Artemisia biennis	Percent Cover	0.10	0.541
	Presence/Absence	0.10	0.549
Artemisia campestris	Percent Cover	0.10	0.545
	Presence/Absence	0.10	0.545
Artemisia longifolia	Percent Cover	0.14	0.414
	Presence/Absence	0.14	0.415
Artemisia ludoviciana	Percent Cover	0.35	0.023
	Presence/Absence	0.34	0.030
Asparagaceae family	Percent Cover	-0.09	0.353
	Presence/Absence	-0.09	0.353
	Proportion of Richness	-0.09	0.353
Asteraceae family	Percent Cover	-0.02	0.915
	Presence/Absence	0.02	0.822
	Proportion of Richness	0.06	0.761
Atriplex prostrata	Percent Cover	0.11	0.859
	Presence/Absence	0.11	0.859
Beckmannia syzigachne	Percent Cover	0.33	0.019
	Presence/Absence	0.26	0.097
Bidens cernua	Percent Cover	-0.01	0.925
	Presence/Absence	0.00	0.941
Plant species that are biennial	Percent Cover	0.04	0.568
	Presence/Absence	0.03	0.627
	Proportion of Richness	0.07	0.488
Boraginaceae family	Percent Cover	0.04	0.787
	Presence/Absence	0.03	0.826
	Proportion of Richness	0.03	0.841
Brassicaceae family	Percent Cover	0.06	0.690
	Presence/Absence	0.06	0.696
	Proportion of Richness	0.05	0.736
Plant species considered to be broadleaf emergent by OWES	Percent Cover	0.12	0.490
	Presence/Absence	-0.07	0.600

Metric	Metric Variations	Spearman Rho	p-value
Plant species considered to be broadleaf emergent by OWES	Proportion of Richness	-0.01	0.754
Bromus inermis	Percent Cover	0.17	0.292
	Presence/Absence	0.17	0.295
Calamagrostis canadensis	Percent Cover	-0.25	0.232
	Presence/Absence	-0.26	0.233
Callitriche palustris	Percent Cover	0.05	0.776
	Presence/Absence	0.05	0.776
Calamagrostis stricta	Percent Cover	-0.19	0.538
	Presence/Absence	-0.18	0.549
Caprifoliaceae family	Percent Cover	-0.07	0.618
	Presence/Absence	-0.07	0.618
	Proportion of Richness	-0.07	0.618
Carex aquatilis	Percent Cover	-0.16	0.822
	Presence/Absence	-0.17	0.785
Carex atherodes	Percent Cover	-0.22	0.098
	Presence/Absence	-0.22	0.133
Carex athrostachya	Percent Cover	-0.08	0.545
	Presence/Absence	-0.08	0.545
Carex bebbii	Percent Cover	-0.02	0.949
	Presence/Absence	-0.02	0.919
Carex brevior	Percent Cover	0.09	0.594
	Presence/Absence	0.09	0.594
Carum carvi	Percent Cover	0.16	0.237
	Presence/Absence	0.16	0.237
Carex genus	Percent Cover	-0.25	0.072
	Presence/Absence	-0.19	0.213
	Proportion of Richness	-0.12	0.354
Carex lacustris	Percent Cover	0.16	0.237
	Presence/Absence	0.16	0.237
Carex pellita	Percent Cover	0.09	0.504
	Presence/Absence	0.08	0.617
Carex praegracilis	Percent Cover	0.05	0.777
	Presence/Absence	0.05	0.760
Carex retrorsa	Percent Cover	-0.14	0.335
	Presence/Absence	-0.14	0.335
Carex utriculata	Percent Cover	-0.05	0.723
	Presence/Absence	-0.04	0.778
Caryophyllaceae family	Percent Cover	-0.17	0.258
	Presence/Absence	-0.17	0.252
	Proportion of Richness	-0.16	0.282

Metric	Metric Variations	Spearman Rho	p-value
Cerastium arvense	Percent Cover	0.16	0.237
	Presence/Absence	0.16	0.237
Chamerion angustifolium	Percent Cover	0.14	0.353
	Presence/Absence	0.14	0.353
Chenopodium album	Percent Cover	-0.02	0.906
	Presence/Absence	-0.01	0.909
Chenopodium capitatum	Percent Cover	-0.12	0.412
	Presence/Absence	-0.12	0.412
Cicuta maculata	Percent Cover	-0.18	0.896
	Presence/Absence	-0.18	0.919
Cirsium arvense	Percent Cover	-0.27	0.082
	Presence/Absence	-0.23	0.158
Cirsium vulgare	Percent Cover	-0.05	0.696
	Presence/Absence	-0.05	0.696
Collomia linearis	Percent Cover	-0.14	0.317
	Presence/Absence	-0.14	0.319
Crepis tectorum	Percent Cover	0.11	0.541
	Presence/Absence	0.12	0.520
Cyperaceae family	Percent Cover	-0.06	0.503
	Presence/Absence	0.06	0.620
	Proportion of Richness	0.17	0.358
Deschampsia cespitosa	Percent Cover	-0.18	0.196
	Presence/Absence	-0.18	0.204
Echinochloa crus-galli	Percent Cover	0.30	0.032
	Presence/Absence	0.30	0.033
Eleocharis acicularis	Percent Cover	0.32	0.024
	Presence/Absence	0.31	0.029
Eleocharis palustris	Percent Cover	0.22	0.197
	Presence/Absence	0.17	0.237
Elymus trachycaulus	Percent Cover	-0.12	0.454
	Presence/Absence	-0.15	0.476
Epilobium ciliatum	Percent Cover	-0.04	0.927
	Presence/Absence	-0.05	0.973
Epilobium leptophyllum	Percent Cover	0.11	0.859
	Presence/Absence	0.11	0.859
Epilobium palustre	Percent Cover	-0.12	0.418
	Presence/Absence	-0.09	0.518
Equisetum arvense	Percent Cover	0.19	0.489
	Presence/Absence	0.20	0.476
Equisetum fluviatile	Percent Cover	-0.22	0.140

Metric	Metric Variations	Spearman Rho	p-value
Equisetum fluviatile	Presence/Absence	-0.22	0.140
Equisetum hyemale	Percent Cover	-0.07	0.618
	Presence/Absence	-0.07	0.618
Equisetaceae family	Percent Cover	-0.24	0.128
	Presence/Absence	-0.25	0.144
	Proportion of Richness	-0.27	0.096
Equisetum palustre	Percent Cover	-0.24	0.603
	Presence/Absence	-0.23	0.629
Equisetum pratense	Percent Cover	-0.23	0.123
	Presence/Absence	-0.23	0.123
Erigeron philadelphicus	Percent Cover	-0.25	0.090
	Presence/Absence	-0.25	0.090
Erysimum cheiranthoides	Percent Cover	0.25	0.090
	Presence/Absence	0.25	0.090
Plants that are eudicots	Percent Cover	-0.14	0.511
	Presence/Absence	-0.13	0.588
	Proportion of Richness	-0.15	0.411
Plants considered to be exotic by AWRAC	Percent Cover	0.00	0.944
	Presence/Absence	0.19	0.300
	Proportion of Richness	0.22	0.252
Plants considered to be exotic and annuals or biennials[4]	Percent Cover	0.12	0.254
	Presence/Absence	0.19	0.160
	Proportion of Richness	0.24	0.080
Graminoid species considered to be exotic	Percent Cover	0.30	0.099
	Presence/Absence	0.31	0.083
	Proportion of Richness	0.32	0.063
Perennial species considered to be exotic	Percent Cover	-0.01	0.809
	Presence/Absence	0.04	0.929
	Proportion of Richness	0.08	0.974
Fabaceae family	Percent Cover	0.00	0.626
	Presence/Absence	-0.02	0.694
	Proportion of Richness	0.00	0.648
Plant species that have facultative status for wetlands	Percent Cover	0.00	0.981
	Presence/Absence	0.11	0.487
	Proportion of Richness	0.16	0.423
Plant species that have facultative-upland status for wetlands	Percent Cover	-0.27	0.132
	Presence/Absence	-0.11	0.778
	Proportion of Richness	-0.10	0.749
Plant species that have facultative wetland status for wetlands	Percent Cover	-0.04	0.968
	Presence/Absence	-0.17	0.377

Metric	Metric Variations	Spearman Rho	p-value
Plant species that have facultative wetland status for wetlands	Proportion of Richness	-0.14	0.436
Fagopyrum esculentum	Percent Cover	0.15	0.300
	Presence/Absence	0.15	0.300
Fallopia convolvulus	Percent Cover	0.19	0.211
	Presence/Absence	0.19	0.208
Fallopia scandens	Percent Cover	0.01	0.972
	Presence/Absence	0.01	0.972
Festuca saximontana	Percent Cover	0.16	0.237
	Presence/Absence	0.16	0.237
Plant species considered to be floating-leaved plants by OWES	Percent Cover	-0.08	0.508
	Presence/Absence	-0.08	0.504
	Proportion of Richness	-0.08	0.500
Plant species that have the forb growth habit according to Moss & Packer, 1983	Percent Cover	0.07	0.546
	Presence/Absence	-0.07	0.825
	Proportion of Richness	0.03	0.844
Fragaria vesca	Percent Cover	-0.19	0.199
	Presence/Absence	-0.20	0.164
Fragaria virginiana	Percent Cover	-0.09	0.353
	Presence/Absence	-0.09	0.353
Plant species considered to be free-floating by OWES	Percent Cover	-0.27	0.050
	Presence/Absence	-0.26	0.061
	Proportion of Richness	-0.24	0.082
Galeopsis tetrahit	Percent Cover	-0.24	0.221
	Presence/Absence	-0.23	0.241
Galium trifidum	Percent Cover	-0.21	0.150
	Presence/Absence	-0.24	0.105
Geum aleppicum	Percent Cover	-0.40	0.005
	Presence/Absence	-0.40	0.005
Geum macrophyllum	Percent Cover	-0.01	0.466
	Presence/Absence	-0.01	0.436
Geum rivale	Percent Cover	0.16	0.237
	Presence/Absence	0.16	0.237
Plant species considered to be sensitive in the Grassland Natural Region of Alberta	Percent Cover	-0.12	0.577
	Presence/Absence	-0.37	0.010
	Proportion of Richness	-0.34	0.012
Plant species considered to be disturbance tolerant in the Grassland Natural Region of Alberta	Percent Cover	0.11	0.493
	Presence/Absence	0.29	0.089
	Proportion of Richness	0.46	0.008
Glyceria borealis	Percent Cover	-0.09	0.503
	Presence/Absence	-0.08	0.529

Metric	Metric Variations	Spearman Rho	p-value
Glyceria grandis	Percent Cover	-0.05	0.800
	Presence/Absence	-0.08	0.966
Glyceria striata	Percent Cover	-0.13	0.800
	Presence/Absence	-0.13	0.780
Plant species that have the graminoid growth habit according to Moss & Packer, 1983	Percent Cover	-0.03	0.577
	Presence/Absence	0.10	0.402
	Proportion of Richness	0.16	0.460
Gratiola neglecta	Percent Cover	0.26	0.063
	Presence/Absence	0.26	0.062
Grindelia squarrosa	Percent Cover	0.02	0.907
	Presence/Absence	0.02	0.879
Grossulariaceae family	Percent Cover	-0.16	0.626
	Presence/Absence	-0.15	0.647
	Proportion of Richness	-0.15	0.669
Plant species considered to be groundcover by OWES	Percent Cover	0.00	0.829
	Presence/Absence	-0.05	0.964
	Proportion of Richness	0.06	0.623
Plant species considered to be hardwood by OWES	Percent Cover	-0.18	0.896
	Presence/Absence	-0.18	0.919
	Proportion of Richness	-0.18	0.896
Hippuris vulgaris	Percent Cover	0.12	0.945
	Presence/Absence	0.12	0.939
Hordeum jubatum	Percent Cover	0.19	0.262
	Presence/Absence	0.24	0.084
Hordeum vulgare	Percent Cover	-0.02	0.804
	Presence/Absence	-0.02	0.804
Plant species whose primary means of propagule dispersal is Hydrochory	Percent Cover	-0.06	0.718
	Presence/Absence	-0.14	0.495
	Proportion of Richness	-0.10	0.384
Inverse Simpson’s diversity index	Value	-0.08	0.858
Evenness	Value	0.12	0.471
Jost Shannon’s diversity	Value	-0.07	0.943
Jost Simpson’s diversity	Value	-0.08	0.858
Juncus balticus	Percent Cover	0.18	0.184
	Presence/Absence	0.17	0.206
Juncaceae family	Percent Cover	0.18	0.181
	Presence/Absence	0.18	0.183
	Proportion of Richness	0.18	0.206
Juncaginaceae family	Percent Cover	-0.09	0.353
	Presence/Absence	-0.09	0.353

Metric	Metric Variations	Spearman Rho	p-value
Juncaginacea family	Proportion of Richness	-0.09	0.353
Juncus vaseyi	Percent Cover	0.16	0.237
	Presence/Absence	0.16	0.237
Krascheninnikovia lanata	Percent Cover	0.04	0.891
	Presence/Absence	0.04	0.879
Lactuca serriola	Percent Cover	0.07	0.715
	Presence/Absence	0.07	0.703
Lamiaceae family	Percent Cover	-0.33	0.014
	Presence/Absence	-0.38	0.005
	Proportion of Richness	-0.33	0.016
Lathyrus othroleucus	Percent Cover	-0.25	0.090
	Presence/Absence	-0.25	0.090
Lemna minor	Percent Cover	-0.27	0.050
	Presence/Absence	-0.26	0.061
Plant species considered to be low shrub by OWES	Percent Cover	-0.15	0.491
	Presence/Absence	-0.16	0.445
	Proportion of Richness	-0.16	0.461
Lycopus asper	Percent Cover	-0.12	0.397
	Presence/Absence	-0.11	0.415
Lysimachia thyrsiflora	Percent Cover	-0.21	0.154
	Presence/Absence	-0.20	0.158
Maianthemum stellatum	Percent Cover	-0.09	0.353
	Presence/Absence	-0.09	0.353
Malva neglecta	Percent Cover	0.24	0.105
	Presence/Absence	0.24	0.105
Malvaceae family	Percent Cover	0.24	0.105
	Presence/Absence	0.24	0.105
	Proportion of Richness	0.24	0.105
Melilotus albus	Percent Cover	-0.02	0.804
	Presence/Absence	-0.02	0.804
Mentha arvensis	Percent Cover	-0.24	0.082
	Presence/Absence	-0.25	0.053
Plants that are monocots	Percent Cover	0.02	0.798
	Presence/Absence	0.09	0.391
	Proportion of Richness	0.22	0.203
Muhlenbergia richardsonis	Percent Cover	0.12	0.934
	Presence/Absence	0.12	0.939
Plant species with multiple methods of dispersal	Percent Cover	0.07	0.707
	Presence/Absence	0.11	0.329
	Proportion of Richness	0.20	0.208

Metric	Metric Variations	Spearman Rho	p-value
Plant species considered to be narrow-leaved emergent by OWES	Percent Cover	-0.02	0.590
	Presence/Absence	0.08	0.459
	Proportion of Richness	0.14	0.484
Plant species native to Alberta	Percent Cover	-0.09	0.535
	Presence/Absence	-0.12	0.657
	Proportion of Richness	-0.15	0.553
Native plant species that are annuals or biennials	Percent Cover	0.26	0.064
	Presence/Absence	0.32	0.032
	Proportion of Richness	0.40	0.011
Native plant species with that are graminoid	Percent Cover	-0.07	0.422
	Presence/Absence	0.08	0.448
	Proportion of Richness	0.15	0.497
Native plant species that are perennials	Percent Cover	-0.23	0.094
	Presence/Absence	-0.26	0.141
	Proportion of Richness	-0.42	0.013
Plant species that are nitrogen fixing	Percent Cover	0.00	0.626
	Presence/Absence	-0.02	0.694
	Proportion of Richness	0.00	0.648
Plant species considered noxious weeds by AWRAC	Percent Cover	-0.14	0.364
	Presence/Absence	-0.20	0.201
	Proportion of Richness	-0.18	0.233
Plant species that have obligate or facultative-wet status for wetlands	Percent Cover	-0.03	0.687
	Presence/Absence	-0.19	0.313
	Proportion of Richness	-0.09	0.460
Plant species that have obligate status for wetlands	Percent Cover	0.06	0.914
	Presence/Absence	-0.14	0.477
	Proportion of Richness	0.01	0.871
Onagraceae family	Percent Cover	-0.06	0.693
	Presence/Absence	-0.02	0.828
	Proportion of Richness	-0.03	0.886
Orchidaceae family	Percent Cover	-0.09	0.353
	Presence/Absence	-0.09	0.353
	Proportion of Richness	-0.09	0.353
Penstemon procerus	Percent Cover	0.10	0.545
	Presence/Absence	0.10	0.545
Persicaria amphibia	Percent Cover	-0.11	0.359
	Presence/Absence	-0.12	0.286
Plant species that are perennial	Percent Cover	-0.25	0.074
	Presence/Absence	-0.23	0.196
	Proportion of Richness	-0.38	0.013

Metric	Metric Variations	Spearman Rho	p-value
Persicaria lapathifolia	Percent Cover	0.39	0.006
	Presence/Absence	0.38	0.009
Petasites frigidus	Percent Cover	-0.42	0.003
	Presence/Absence	-0.42	0.003
Phalaris arundinacea	Percent Cover	-0.08	0.586
	Presence/Absence	-0.11	0.460
Phleum pratense	Percent Cover	0.20	0.464
	Presence/Absence	0.20	0.476
Plant species considered to be sensitive in the Parkland Natural Region of Alberta	Percent Cover	-0.26	0.082
	Presence/Absence	-0.22	0.192
	Proportion of Richness	-0.24	0.138
Plant species considered to be disturbance tolerant in the Parkland Natural Region of Alberta	Percent Cover	-0.04	0.926
	Presence/Absence	0.18	0.164
	Proportion of Richness	0.32	0.044
Platanthera hyperborea	Percent Cover	-0.09	0.353
	Presence/Absence	-0.09	0.353
Plantago major	Percent Cover	0.04	0.887
	Presence/Absence	0.04	0.879
Plantaginaceae family	Percent Cover	0.15	0.528
	Presence/Absence	0.17	0.456
	Proportion of Richness	0.18	0.377
Plagiobothrys scouleri	Percent Cover	0.04	0.787
	Presence/Absence	0.03	0.826
Poaceae family	Percent Cover	0.01	0.952
	Presence/Absence	0.09	0.465
	Proportion of Richness	0.20	0.295
Poa palustris	Percent Cover	0.13	0.549
	Presence/Absence	0.09	0.722
Poa pratensis	Percent Cover	-0.17	0.246
	Presence/Absence	-0.20	0.152
Polygonum aviculare	Percent Cover	0.16	0.626
	Presence/Absence	0.15	0.646
Polemoniaceae	Percent Cover	-0.14	0.317
	Presence/Absence	-0.14	0.319
	Proportion of Richness	-0.14	0.317
Polygonum ramosissimum	Percent Cover	-0.17	0.225
	Presence/Absence	-0.17	0.219
Polygonaceae family	Percent Cover	0.25	0.104
	Presence/Absence	0.22	0.200
	Proportion of Richness	0.23	0.172

Metric	Metric Variations	Spearman Rho	p-value
Populus tremuloides	Percent Cover	-0.18	0.896
	Presence/Absence	-0.18	0.919
Potamogetonaceae family	Percent Cover	-0.08	0.508
	Presence/Absence	-0.09	0.488
	Proportion of Richness	-0.09	0.469
Potentilla anserina	Percent Cover	-0.08	0.363
	Presence/Absence	-0.08	0.345
Potamogeton gramineus	Percent Cover	0.05	0.776
	Presence/Absence	0.05	0.776
Potentilla norvegica	Percent Cover	0.26	0.104
	Presence/Absence	0.26	0.106
Potamogeton richardsonii	Percent Cover	-0.14	0.298
	Presence/Absence	-0.14	0.295
Potentilla rivalis	Percent Cover	0.13	0.401
	Presence/Absence	0.12	0.415
Primulaceae family	Percent Cover	-0.21	0.154
	Presence/Absence	-0.20	0.158
	Proportion of Richness	-0.21	0.154
Ranunculus cymbalaria	Percent Cover	-0.09	0.997
	Presence/Absence	-0.10	0.973
Ranunculus gmelinii	Percent Cover	0.03	0.713
	Presence/Absence	0.02	0.675
Ranunculus macounii	Percent Cover	-0.05	0.653
	Presence/Absence	-0.05	0.690
Ranunculus sceleratus	Percent Cover	-0.23	0.114
	Presence/Absence	-0.23	0.118
Ranunculaceae family	Percent Cover	-0.14	0.300
	Presence/Absence	-0.19	0.234
	Proportion of Richness	-0.19	0.200
Plant species considered rare in Alberta	Percent Cover	-0.03	0.865
	Presence/Absence	-0.01	0.960
	Proportion of Richness	0.01	0.934
Plant species found to have recalcitrant litter decomposition	Percent Cover	0.21	0.129
	Presence/Absence	0.15	0.459
	Proportion of Richness	0.21	0.300
Ribes oxyacanthoides	Percent Cover	-0.16	0.626
	Presence/Absence	-0.15	0.647
Plant species considered to be robust emergent by OWES	Percent Cover	0.01	0.901
	Presence/Absence	0.05	0.749
	Proportion of Richness	0.08	0.600

Metric	Metric Variations	Spearman Rho	p-value
Rorippa palustris	Percent Cover	0.16	0.268
	Presence/Absence	0.15	0.302
Rosaceae family	Percent Cover	0.16	0.199
	Presence/Absence	0.08	0.348
	Proportion of Richness	0.12	0.212
Rosa acicularis	Percent Cover	-0.23	0.211
	Presence/Absence	-0.24	0.199
Rubiaceae family	Percent Cover	-0.21	0.150
	Presence/Absence	-0.24	0.105
	Proportion of Richness	-0.23	0.124
Rubus pubescens	Percent Cover	-0.09	0.353
	Presence/Absence	-0.09	0.353
Rumex britannica	Percent Cover	-0.19	0.184
	Presence/Absence	-0.19	0.184
Rumex crispus	Percent Cover	0.10	0.819
	Presence/Absence	0.08	0.932
Rumex fueginus	Percent Cover	0.37	0.011
	Presence/Absence	0.38	0.009
Rumex occidentalis	Percent Cover	-0.24	0.682
	Presence/Absence	-0.24	0.686
Rumex salicifolius	Percent Cover	-0.10	0.464
	Presence/Absence	-0.11	0.419
Salix discolor	Percent Cover	-0.25	0.090
	Presence/Absence	-0.25	0.090
Salix exigua	Percent Cover	-0.35	0.138
	Presence/Absence	-0.35	0.150
Salicaceae family	Percent Cover	-0.36	0.092
	Presence/Absence	-0.36	0.115
	Proportion of Richness	-0.37	0.104
Salix lasiandra	Percent Cover	-0.20	0.162
	Presence/Absence	-0.20	0.158
Salix lucida	Percent Cover	-0.09	0.353
	Presence/Absence	-0.09	0.353
Salix pseudomonticola	Percent Cover	-0.09	0.353
	Presence/Absence	-0.09	0.353
Salicornia rubra	Percent Cover	-0.05	0.696
	Presence/Absence	-0.05	0.696
Schoenoplectus acutus	Percent Cover	-0.02	0.804
	Presence/Absence	-0.02	0.804
Schoenoplectus pungens	Percent Cover	0.15	0.348

Metric	Metric Variations	Spearman Rho	p-value
Schoenoplectus pungens	Presence/Absence	0.15	0.354
Schoenoplectus tabernaemontani	Percent Cover	0.13	0.274
	Presence/Absence	0.14	0.285
Scirpus microcarpus	Percent Cover	-0.22	0.140
	Presence/Absence	-0.22	0.140
Scolochloa festucacea	Percent Cover	-0.21	0.072
	Presence/Absence	-0.17	0.122
Scutellaria galericulata	Percent Cover	-0.53	0.000
	Presence/Absence	-0.52	0.000
Shannon's diversity index	Value	-0.07	0.943
Plant species that have the shrub growth habit according to Moss & Packer, 1983	Percent Cover	-0.32	0.168
	Presence/Absence	-0.31	0.215
	Proportion of Richness	-0.32	0.185
Simpson's diversity index	Value	-0.08	0.858
Sium suave	Percent Cover	-0.16	0.272
	Presence/Absence	-0.15	0.301
Solidago altissima	Percent Cover	-0.23	0.482
	Presence/Absence	-0.23	0.501
Sonchus arvensis	Percent Cover	-0.12	0.341
	Presence/Absence	-0.15	0.268
Sonchus asper	Percent Cover	-0.13	0.888
	Presence/Absence	-0.14	0.854
Sonchus oleraceus	Percent Cover	0.19	0.161
	Presence/Absence	0.19	0.161
Spergularia salina	Percent Cover	-0.17	0.225
	Presence/Absence	-0.17	0.219
Stachys pilosa	Percent Cover	-0.30	0.033
	Presence/Absence	-0.29	0.034
Stellaria longifolia	Percent Cover	-0.25	0.090
	Presence/Absence	-0.25	0.090
Suaeda calceoliformis	Percent Cover	-0.05	0.696
	Presence/Absence	-0.05	0.696
Symphyotrichum boreale	Percent Cover	-0.08	0.524
	Presence/Absence	-0.08	0.525
Symphyotrichum ericoides	Percent Cover	-0.19	0.184
	Presence/Absence	-0.19	0.184
Symphyotrichum lanceolatum	Percent Cover	-0.05	0.825
	Presence/Absence	-0.03	0.978
Symphoricarpos occidentalis	Percent Cover	-0.07	0.618
	Presence/Absence	-0.07	0.618

Metric	Metric Variations	Spearman Rho	p-value
Symphyotrichum puniceum	Percent Cover	-0.16	0.311
	Presence/Absence	-0.16	0.282
Plant species considered to be tall shrubs by OWES	Percent Cover	-0.42	0.037
	Presence/Absence	-0.42	0.047
	Proportion of Richness	-0.42	0.048
Taraxacum officinale	Percent Cover	0.30	0.128
	Presence/Absence	0.29	0.145
Tephroseris palustris	Percent Cover	0.23	0.140
	Presence/Absence	0.23	0.140
Thlaspi arvense	Percent Cover	0.15	0.294
	Presence/Absence	0.14	0.337
Tragopogon dubius	Percent Cover	0.08	0.677
	Presence/Absence	0.08	0.684
Plant species considered to be trees by OWES	Percent Cover	-0.18	0.896
	Presence/Absence	-0.18	0.919
	Proportion of Richness	-0.18	0.896
Plant species considered to be trees or shrubs by OWES	Percent Cover	-0.32	0.160
	Presence/Absence	-0.31	0.228
	Proportion of Richness	-0.32	0.185
Trifolium hybridum	Percent Cover	0.28	0.056
	Presence/Absence	0.28	0.055
Triglochin maritima	Percent Cover	-0.09	0.353
	Presence/Absence	-0.09	0.353
Typhaceae family	Percent Cover	0.01	0.796
	Presence/Absence	0.03	0.870
	Proportion of Richness	0.04	0.973
Typha latifolia	Percent Cover	0.01	0.796
	Presence/Absence	0.03	0.870
Plant species that have upland status for wetlands	Percent Cover	0.28	0.048
	Presence/Absence	0.22	0.087
	Proportion of Richness	0.30	0.039
Urtica dioica	Percent Cover	-0.35	0.016
	Presence/Absence	-0.35	0.016
Urticaceae family	Percent Cover	-0.35	0.016
	Presence/Absence	-0.35	0.016
	Proportion of Richness	-0.35	0.016
Plant species that exhibit vegetative reproduction	Percent Cover	-0.17	0.179
	Presence/Absence	-0.24	0.152
	Proportion of Richness	-0.32	0.023
Veronica scutellata	Percent Cover	-0.15	0.292

Metric	Metric Variations	Spearman Rho	p-value
Veronica scutellata	Presence/Absence	-0.15	0.295
Vicia americana	Percent Cover	-0.20	0.527
	Presence/Absence	-0.20	0.556
Plant species that have the vine growth habit according to Moss & Packer, 1983	Percent Cover	-0.02	0.814
	Presence/Absence	-0.02	0.741
	Proportion of Richness	0.01	0.663
Viola adunca	Percent Cover	-0.09	0.353
	Presence/Absence	-0.09	0.353
Violaceae family	Percent Cover	-0.09	0.353
	Presence/Absence	-0.09	0.353
	Proportion of Richness	-0.09	0.353
Plant species identified as a weed in the prairie provinces of Canada by Bubar et al., 2000	Percent Cover	0.09	0.626
	Presence/Absence	0.13	0.358
	Proportion of Richness	0.29	0.087
Plant species whose primary means of propagule dispersal is Zoochory	Percent Cover	0.15	0.449
	Presence/Absence	0.06	0.494
	Proportion of Richness	0.15	0.344

Appendix 3.1: All FRAGSTATS metrics.

Table A.3.1.1: All FRAGSTATS metrics available at the patch level including the metric type, description and units where applicable.

Metric Type	Metric	Code	Description	Units	Additional Requirements
Area/Edge	Area	AREA	Area of each patch in the file	ha	
Area/Edge	Perimeter	PERIM	Perimeter of the patch including any internal holes	m	
Area/Edge	Radius of Gyration	GYRATE	Mean distance between each cell in the patch and the patch centroid	m	
Shape	Perimeter-Area Ratio	PARA	Ratio of patch perimeter to area. A simple measure of complexity	--	
Shape	Shape Index	SHAPE	Patch perimeter divided by the square root of patch area adjusted by a constant	--	
Shape	Fractal Dimension Index	FRAC	2 times the logarithm of patch perimeter divided by the logarithm of patch area.	--	
Shape	Related Circumscribing Circle	CIRCLE	1 minus patch area divided by the area of the smallest circumscribing circle	--	
Shape	Contiguity Index	CONTIG	Average contiguity values (i.e. sum of the cell values divided by the total # of pixels in the patch minus 1, divided by the sum of the template values minus 1).	--	
Core Area	Core Area	CORE	Area within the patch that is further than the specified depth-of-edge distance from the patch perimeter	ha	Edge Depth Table
Core Area	Number of Core Areas	NCORE	Number of Core Areas contained within the patch boundary	--	Edge Depth Table
Core Area	Core Area Index	CAI	Patch core area divided by total patch area	%	Edge Depth Table
Contrast	Edge Contrast Index	ECON	Sum of the patch perimeter segments multiplied by their corresponding contrast weights divided by total patch perimeter	%	Edge Contrast Table

Metric Type	Metric	Code	Description	Units	Additional Requirements
Aggregation	Euclidean Nearest-Neighbor Distance	ENN	Distance to the nearest neighboring patch of the same type based on the shortest edge-to-edge distance	m	
Aggregation	Proximity Index	PROX	Sum of patch area divided by the nearest edge-to-edge distance squared between the patch and the focal patch of all patches of the same type whose edges are within a specified distance of the focal patch	--	Search radius
Aggregation	Similarity Index	SIMI	Sum over all neighboring patches with edges within a specified distance of the focal patch, of neighboring patch area times a similarity coefficient between the focal patch type	--	Similarity Table, Search Radius

Table A.3.1.2: All FRAGSTATS metrics available at the class level including the metric type, description and units where applicable.

Metric Type	Metric	Code	Description	Units	Additional Requirements
Area/Edge	Total Area	CA/TA	Sum of the areas of all patches of the same class	ha	
Area/Edge	Percentage of the Landscape	PLAND	Sum of the areas of all patches of the same class, divided by the total landscape area (proportional abundance)	%	
Area/Edge	Largest Patch Index	LPI	Percentage of the landscape covered by the largest patch	%	
Area/Edge	Total Edge	TE	Sum of the lengths of all edge segments for each patch type	m	
Area/Edge	Edge Density	ED	Sum of the lengths of all edge segments for each patch type divided by the total landscape area	m/ha	
Shape	Perimeter-Area Fractal Dimension	PAFRAC	2 divided by the slope of regression line obtained by regressing the logarithm of patch area against the logarithm of patch perimeter	--	
Core Area	Total Core Area	TCA	Sum of core areas of each patch of patch type	ha	
Core Area	Core Area Percentage of Landscape	CPLAND	Sum of core areas for each patch divided by the total landscape area	%	
Core Area	Number of Disjunct Core Areas	NDCA	Number of disjunct core areas contained within each patch of the corresponding type	--	
Core Area	Disjunct Core Area Density	DCAD	Sum of the disjunct core areas contained within each patch type divided by total landscape area	#/ha	Edge Depth Table
Contrast	Contrast-Weighted Edge Density	CWED	Sum of the lengths of each edge segment per patch type, multiplied by the corresponding contrast weight divided by total landscape area	m/ha	Edge Contrast Table
Contrast	Total Edge Contrast Index	TECI	Sum of the lengths of each edge segment per patch type, multiplied by the corresponding contrast weight	%	Edge Contrast Table

Metric Type	Metric	Code	Description	Units	Additional Requirements
			divided the lengths of all edge edges of the same type		
Aggregation	Interspersion and Juxtaposition Index	IJI	Sum of the length of each unique edge type involving the corresponding patch type divided by the total length of edge involving the same type, multiplied by the logarithm of the same quantity summed over each unique edge the divided by the logarithm of the number of patch types minus 1	%	
Aggregation	Percentage of Like Adjacencies	PLADJ	Number of like adjacencies involving the focal class divided by the total number of cell adjacencies involving the focal class	%	
Aggregation	Aggregation Index	AI	Number of like adjacencies involving the corresponding class, divided by the maximum possible number of like adjacencies involving the corresponding class.	%	
Aggregation	Clumpiness Index	CLUMPY	Proportional deviation of the proportion of like adjacencies involving the corresponding class from that expected under a spatially random distribution.	%	
Aggregation	Landscape Shape Index	LSI	.25 the sum of the entire landscape boundary and all edge segments within the landscape boundary involving the corresponding patch type divided by the square root of the total landscape area	--	
Aggregation	Normalized Landscape Shape Index	NLSI	Total length of edge of the corresponding class given in number of cell surfaces minus the minimum length of class edge possible for a maximally aggregated class	--	

Metric Type	Metric	Code	Description	Units	Additional Requirements
Aggregation	Patch Cohesion Index	COHESION	1 minus the sum of patch perimeter divided by the sum of patch perimeter times the square root of patch area divided by 1 minus 1 over the square root of the total number of cells in the landscape	%	
Aggregation	Number of Patches	NP	Number of patches of the corresponding type	--	
Aggregation	Patch Density	PD	Number of patches of the corresponding type divided by the total landscape area	m/ha	
Aggregation	Landscape Division Index	DIVISION	1 minus the sum of patch area divided by total landscape area, quantity squared, summed across all patches of the corresponding patch type	proportion	
Aggregation	Splitting Index	SPLIT	Total landscape area squared divided by the sum of patch area squared summed across all patches of all the corresponding patch type	--	
Aggregation	Effective Mesh Size	MESH	sum of patch area squared, summed across all patches of the corresponding patch type, divided by total landscape area	ha	
Aggregation	Connectance	CONNECT	number of functional joinings between all patches of the corresponding patch type divided by the total number of possible joinings between all patches of the corresponding patch type	%	

Table A3.1.3: All FRAGSTATS metrics available at the landscape level including the metric type, description and units where applicable.

Metric Type	Metric	Code	Description	Units	Additional Requirements
Area/Edge	Total Area	TA	Total area of the landscape	ha	
Area/Edge	Largest Patch Index	LPI	Area of the largest patch in the landscape divided by the total landscape area	%	
Area/Edge	Total Edge	TE	Sum of the lengths of all edge segments in the landscape	m	
Area/Edge	Edge Density	ED	Sum of the lengths of all edge segments in the landscape divided by the total landscape area	m/ha	
Shape	Perimeter-Area Fractal Dimension	PAFRAC	2 divided by the slope of regression line obtained by regressing the logarithm of patch area against the logarithm of patch perimeter	--	
Core Area	Total Core Area	TCA	Sum of core area of each patch	ha	Edge Depth Table
Core Area	Number of Disjunct Core Areas	NDCA	Sum of the number of disjunct core areas contained within each patch in the landscape	--	Edge Depth Table
Core Area	Disjunct Core Area Density	DCAD	Sum of the number of disjunct core areas contained within each patch in the landscape divided by total landscape area	#/100 ha	Edge Depth Table
Contrast	Contrast-Weighted Edge Density	CWED	Sum of the lengths of each edge segment in the landscape multiplied by the contrast weight divided by the total landscape area	m/ha	Edge Contrast Table
Contrast	Total Edge Contrast Index	TECI	Sum of the lengths of each edge segment in the landscape multiplied by the contrast weight divided by the total length of edge in the landscape	%	Edge Contrast Table

Metric Type	Metric	Code	Description	Units	Additional Requirements
Aggregation	Contagion Index	CONTAG	Minus the sum of the proportional abundance of each patch type multiplied by the proportion of adjacencies between cells of that patch type and another patch type, multiplied by the logarithm of the same quantity summed over each unique adjacency type and each patch type, divided by 2 times the logarithm of the number of patch types.	%	
Aggregation	Interspersion and Juxtaposition Index	IJI	Sum of the length of each unique edge type divided by the total landscape edge, multiplied by the logarithm of the same quantity summed over each unique edge the divided by the logarithm of the number of patch types minus 1, divided by 2	%	
Aggregation	Percentage of Like Adjacencies	PLADJ	Sum of the number of like adjacencies for each patch type, divided by the total number of cell adjacencies in the landscape	%	
Aggregation	Aggregation Index	AI	Number of like adjacencies involving the corresponding class, divided by the max possible number of like adjacencies involving the corresponding class, summed over all classes	%	
Aggregation	Landscape Shape Index	LSI	0.25 times the sum of the entire landscape boundary and all edge segments within the landscape boundary divided by the square root of the total landscape area.	--	

Metric Type	Metric	Code	Description	Units	Additional Requirements
Aggregation	Patch Cohesion Index	COHESION	1 minus the sum of patch perimeter divided by the sum of patch perimeter times the square root of patch area for all patches in the landscape divided by 1 minus 1 over the square root of the total number of cells in the landscape	%	
Aggregation	Number of Patches	NP	Equals the number of patches in the landscape	--	
Aggregation	Patch Density	PD	Number of patches in the landscape divided by total landscape area	#/100 ha	
Aggregation	Landscape Division Index	DIVISION	1 minus the sum of patch area divided by total landscape area, quantity squared, summed across all patches in the landscape	proportion	
Aggregation	Splitting Index	SPLIT	total landscape area squared, divided by the sum of patch area squared, summed across all patches in the landscape	--	
Aggregation	Effective Mesh Size	MESH	1 divided by total landscape area, multiplied by the sum patch area squared, summed across all patches in the landscape	ha	
Aggregation	Connectance Index	CONNECT	number of functional joinings between all patches of the same patch type, divided by the total number of possible joinings between all patches of the same type	%	
Diversity	Patch Richness	PR	Number of different patch types present in the landscape	--	
Diversity	Patch Richness Density	PRD	Number of different patch types present in the landscape divided by total landscape area	#/100ha	
Diversity	Relative Patch Richness	RPR	Number of different patch types present within the landscape boundary divided by the maximum potential number of patch types specified by the user	%	

Metric Type	Metric	Code	Description	Units	Additional Requirements
Diversity	Shannon's Diversity Index	SHDI	minus the sum across all patch types of the proportional abundance of each patch type multiplied by that proportion	--	
Diversity	Simpson's Diversity Index	SIDI	1 minus the sum, across all patches, of the proportional abundance of each patch type squared	--	
Diversity	Modified Simpson's Diversity Index	MSIDI	minus the log of the sum, across all patch types, of the proportional abundance of each patch type squared	--	
Diversity	Shannon's Evenness Index	SHEI	minus the sum, across all patch types, of the proportional abundance of each patch type multiplied by that proportion divided by the log of the number of patch types.	--	
Diversity	Simpson's Evenness Index	SIEI	1 minus the sum, across all patches, of the proportional abundance of each patch type squared, divided by 1 minus the number of patch types	--	
Diversity	Modified Simpson's Evenness Index	MSIEI	minus the log of the sum, across all patch types, of the proportional abundance of each patch type squared, divided by the log of the number of patch types	--	

Appendix 3.2: Quantifying Data Uncertainty

Overview of errors in landscape ecology, a paper written to identify and quantify the error and uncertainty in our mapping procedure. This paper was originally submitted to Dr. D. Robinson as part of a course credit.

Landscape ecology provides a means to quantify the relationship between discrete habitats within a broad landscape, and relate them to the biological processes within that landscape (Bell, Fonseca, & Motten, 1997; Turner, 1989). This is frequently performed using a combination of remote sensing and GIS to derive patterns and environmental metrics that can provide an understanding about the landscape and quantify landscape structure (McGarigal & Marks, 1994). This information is valuable for studying landscapes but is also applicable to regional management, conservation, land use planning and restoration (Wu, 2006). While remote sensing and GIS are extremely useful and relatively easy tools for simplifying a complex landscape, there is the potential for error and uncertainty to muddy the results of these tools (MacEachren et al., 2005). Any error introduced into the acquisition of landscape data will compound throughout analyses and lead to inaccurate final results (Gahegan & Ehlers, 2000). Therefore, care must be taken to monitor and reduce error at every step of the process when performing a landscape ecology analysis. This paper will address the areas where error is introduced into landscape ecology methodologies and, through use of a case study, demonstrate steps that can be taken to identify, visualize, and quantify this error and uncertainty.

A.3.2.1 Error and uncertainty

A.3.2.1.1 Types of Error

For our purposes we will consider error as the quantifiable deviation of a value from the actual value (MacEachren et al., 2005; McKenzie et al., 2015). Sources of error can be introduced through equipment limitations, data or formatting mistakes, user error, or environmental factors that impede the accuracy or precision of the work (DiBiase, 2014; Wormley, 2010). It is important to recognize the various types of error that can occur throughout the process, in order to minimize their effect within a dataset.

A.3.2.1.2 Equipment error

Equipment error often occurs when working with data that is derived using some sort of mechanical device, such as a Global Positioning System (GPS) device. Error in GPS devices can be introduced in a number of ways, namely: if there is a discrepancy between the clock in the GPS receiver and the satellite clock, if there are insufficient satellites in orbit above one's location, or as a result of atmospheric conditions affecting the signal as it travels between the satellites and the receiver (DiBiase, 2014; Wormley, 2010). Most sources of equipment error are compensated for with ground stations that calculate error corrections based on atmospheric conditions and clock drift, and transmit this information to the satellites in order to reduce the error seen at the receiver – however, some error still occurs (DiBiase, 2014; Wormley, 2010). A common type of error is multipath error, which refers to error caused when objects on the ground, such as trees and buildings, interfere with the signal reaching the GPS receiver. Multipath error can be minimized by placement of the GPS receiver. Despite best efforts, error will ultimately, be present in your final GPS readings and steps should be taken to quantify this error.

A.3.2.1.3 Data and formatting error

Data or formatting error occurs when data sets are being manipulated or altered in some way. In landscape ecology, data error often occurs when converting from vector data to raster data or vice versa (Choudhry & Morad, 1998). Vector data, which is comprised of points, lines, and polygons arranged spatially, can be highly accurate depending on the data source, as it can more precisely match the shape of landscape features than raster data (Heywood et al., 2006; Sırrı Mara, et. al, 2010). The error inherent to vector data is often in the form of geometric error, that is, error in the creation of vector features that can disrupt analyses or misrepresent the portrayed features (Sırrı Mara et al., 2010; Ubeda & Egenhofer, 1997). Errors, such as overlapping features or gaps between neighboring features or polygons that are not entirely closed, can misrepresent the portrayed features and introduce error (Heywood et al., 2006; Sırrı Mara et al., 2010). Raster data, in contrast, is a simple grid with cells of a certain size that have only one value (Congalton, 1997; Heywood et al., 2006). Most remotely sensed data, such as satellite imagery, is presented in raster format as the creation of rasters is easy to automate, compared with vectors which often require more human input (Congalton, 1997). Raster data is easy to create and easy to use, but it is often more generalized and results in a loss of detail that might be preserved if the data was in vector format et. al, 1996). The quality of raster data is largely dependant on the size of the cells, which in turn is dependant on the spatial resolution of the device used to create the raster data (Carver & Brunsdon, 1994).

In GIS, the common belief is that while vector data is of higher quality and provides a more accurate representation of real-world conditions, raster data is favoured for complex analyses because raster files are smaller, simpler, and easier to process (Carver & Brunsdon, 1994; Liao, Bai, & Bai, 2012; Wade et al., 2003). Vector data, such as digitized aerial photography, may require conversion to raster format for analysis. This means that data sets containing spatially arranged points, lines, and polygons will be converted to a cell-grid where each cell contains only one value (Congalton, 1997; Heywood et al., 2006). This type of conversion dilutes the accuracy of the vector data, especially the shape of landscape features (Carver & Brunsdon, 1994). Converting from raster to vector has varied results depending on the raster cell size, but ultimately there will be some discrepancy between the shape of the feature in raster format and the shape in vector format (Congalton, 1997). Data conversion error, like equipment error, is something that will be present and is quantifiable and, like equipment error, an awareness of data and formatting error is important.

A.3.2.1.4 User error and environmental uncertainty

User error and environmental error are much harder to quantify and account for. As with any process that involves human input, user error will always be present to a certain degree (Hales & Pronovost, 2006). Developing a simple, reliable, step-wise protocol for all procedures carried out by people helps to reduce user error and ensure sampling is carried out in the same way every time. Environmental factors can introduce error which often leads to other sources of error, including clouds in satellite imagery, or interference with a GPS signal. However, environmental factors often produce more unquantifiable uncertainty than actual, quantifiable error. While error in measurements is easy to define, quantify, and correct, uncertainty is far less so (MacEachren et al., 2005). Uncertainty refers to a discrepancy between a measured value and the true value, which is not sufficiently clear or definable (Van Leeuwen & Orr, 2006). Uncertainty is not objectively known and can be introduced either when data is being collected or when data is being processed (Gahegan & Ehlers, 2000; MacEachren et al., 2005). It is often easy to conceptualize uncertainty but difficult to actually quantify uncertainty as uncertainty arises when the true value being measured is unknown (Gahegan & Ehlers, 2000; Van Leeuwen & Orr, 2006). It is important to be aware of all types of error and uncertainty when planning and carrying out an analysis, especially as landscape ecology is highly procedural work and error in an earlier step can propagate and lead to increased error further on.

A.3.2.2.0 Error and uncertainty case study

A.3.2.2.1 Study design

The case study presented here focuses on a subset of wetlands located in the Prairie Pothole Region (PPR) of Alberta. These wetlands are being visited as part of a larger project studying the responses of vegetation communities in prairie pothole wetlands to human disturbance. Wetlands in the Prairie Pothole Region (PPR) of Alberta were sampled over three years, and vegetation communities were delineated and mapped in order to create GIS polygons for each wetland. These polygons were used to generate landscape metrics using FRAGSTATS to describe the characteristics of individual vegetation communities in the broader landscape of the entire wetland. In this case study, I present three wetlands that were sampled four times over the course of this project: once in 2014, once in 2015, and twice in 2016. The goal of this case study is to identify the sources of error and uncertainty introduced throughout the sampling and analysis of these wetlands and, if possible, visualize or quantify the error or uncertainty. Error is expected to be introduced through the device used to carry out the mapping, the human technician carrying out the mapping, and the data conversion used while manipulating the final data. Additionally, uncertainty is introduced as a result of the environmental characteristics of the wetland ecosystem, which leads to difficulty in clearly delineating vegetation communities.

A.3.2.2.2 Field methods

Sampling of wetlands took place over two years in Alberta's PPR: 48 wetlands were sampled in 2014, and 72 were sampled in 2015. Some wetlands were visited in both years; 24 of the 72 sites visited in 2015 were retained from the 2014 sampling year. Each wetland's vegetation communities were mapped during peak growing season (late July to August) to ensure accurate identification of the vegetation. For this case study, three of the revisit sites were visited again in 2016 and mapped twice with a span of two weeks between visits.

The creation of wetland vegetation polygons was carried out in the field using an Sx Blue II+ GPS/GNSS receiver manufactured by Geneq Inc., a device worn in a backpack with a 1m telescopic antenna. The receiver connects via Bluetooth to a Juno T41 handheld manufactured by Trimble. ArcPad version 10.0 was installed on the handheld to create and edit vegetation polygons in the field. The receiver connects to both American GPS satellites and Russian GLONASS satellites and can use a satellite-based augmentation system (SBAS) to reduce the error attributed with atmospheric interference, clock drift and uncertainty in satellite position. While capable of using a Differential Global Navigation Satellite System (DGNSS), there are no Differential Global Positioning System (DGPS) base stations within range of our study area. The horizontal accuracy of the receiver is < 2.5m 2dRMS with 95% confidence ("SX Blue II+ GNSS Technical Schematics," 2015). This value is obtained by taking the square root of the average of the squared errors, then multiplying it by two to obtain twice the root mean square error, or 2dRMS. What this means it that, while measuring a point, we are 95% sure that the true location lies within a 2.5m radius circle around our receiver.

To assess the error of our mapping receiver in the field, I compared points recorded by the Geneq device to a Leica CS15 GNSS system manufactured by Leica Geosystems. This system consists of a handheld receiver unit (rover) and a stationary receiver (base station). The base station is placed at a known location such as a benchmark or other control point, while the rover is used to take the desired readings. The base station is constantly taking readings of its own position while in constant contact with the rover, so the base station knows its own position with high accuracy as well as the distance between itself and the rover (Stevens, Smith, & Biancheti, 2012). The error in the base station is calculated as the difference between its readings and the precise location of the control point that the base station is placed at. This error is used as a differential correction for all the measures taken by the rover. The error of this device after differential correction is 10mm + 1ppm RMS with 65% probability ("Leica CS10/CS15 & GS Sensors User Manual," 2014). This lets us say that while measuring a point we are 65% sure that the true location lies within a circle of radius equal to 10mm + (10^{-6}x the distance between the rover and the base station).

To minimize the effect of user error a rigorous, step-wise protocol was created to ensure that the same steps were followed each time a site was mapped. In addition, the same technician was used for every site mapped to ensure consistency, and field notes and photos were taken to provide supplementary information in the case of confusion in the data. For the 2016 sampling period, a twoweek gap was placed between site visits to ensure that any trampled vegetation would be restored and any paths made during the first visit would not be visible, which could result in bias in the technician's mapping. The vegetation polygons were created *in situ* using ArcPad version 10.0 on the Trimble handheld unit. As the technician walked the perimeter of each wetland assemblage, the Trimble created a point every meter. Once the perimeter was completed, the points formed vertices for an enclosed polygon which represented the vegetation assemblage being delineated. Given the inherent error in the GPS receiver, and the general difficulties of walking through wetland terrain, the resulting polygons often had small topology errors such as overlaps and sliver. A technician examined and corrected the errors in each individual polygon to ensure that the topology error was removed before the data was converted to raster format for subsequent analysis.

A.3.2.2.3 Prairie pothole wetlands and uncertainty

Uncertainty in this case study largely stems from the natural characteristics of the ecosystem being studied, specifically the difficulty in distinguishing assemblage boundaries. prairie pothole wetlands are characterized by a distinct pattern of vegetation zonation (Stewart & Kantrud, 1971). This pattern of zonation is a result of a change in the water level along a gradient within these wetlands (Keddy, 1999; Seabloom & van der Valk, 2003b), which is the principal driver of vegetation community establishment within these wetlands (Seabloom & van der Valk, 2003b). Wetland vegetation is characterized by its tolerance to wet conditions, with wetland obligate species occurring in the wettest conditions and a gradual transition to facultative wetland species in drier areas within the wetland until finally transitioning to upland species at the wetland edge (Tiner, 1999). Thus we have an ecocline: a series of heterogeneous vegetation communities that are found along a changing environmental gradient (Attrill & Rundle, 2002). Given the transitional nature of vegetation communities observed along ecoclines, the actual boundary between communities can be difficult to determine.

When attempting to delineate the different vegetation assemblages within a wetland, the transition area between communities can be highly mixed and an exact assemblage boundary is often difficult to identify. This is further complicated by the presence of other ecological gradients besides the hydrologic gradient, such as salinity, which can have an effect on vegetation communities (CEMA, 2014; Keddy, 1999; van der Valk, 1981). Other factors such as disturbance, land cover attributes, and interspecies competition also affect vegetation zonation (Galatowitsch et al., 2000; Keddy, 1999; Seabloom & van der Valk, 2003a). Disturbance such as agriculture and cattle grazing often have a physical impact on wetlands, as farming machinery and cattle are both capable of destroying vegetation. In cases where wetland vegetation is affected by an external disturbance that generates a highly disrupted vegetation community, wetlands are more likely to be colonized by opportunistic plant species, resulting in a vegetation community that is highly mixed. Since disturbance can make it harder to distinguish the boundary of vegetation zones, there is increased uncertainty at sites that have a higher disturbance level.

A.3.2.2.4 Data formatting

The vegetation community polygons for each study year (2014, 2015, 2016) were merged together to form one shape file. The attribute table for this merged shapefile was exported and the community names were compared with vegetation plot data to ensure the correct community names. Once the community names were corrected, each community was assigned an integer value as a numerical ID and a master list of numerical ID's was created. This process was repeated for each year and any new communities were added to the master list. These tables were then loaded into ArcMap and joined with their corresponding year so the numerical ID's could be copied into the attribute tables for the corresponding year. Finally, the merged shapefiles were separated into shapefiles for each site using the Split by Attribute tool. Raster conversion was carried out using a cell-size of 0.5 meters based on the smallest polygons created in the field. Rasters were exported as a Tiff (.tif extension) and imported into Fragstats in this format. The master list of community ID's was saved as a comma delimited text file with an .fcd extension to be used as class descriptor table for Fragstats. A number of metrics was generated using FRAGSTATS.

The FRAGSTATS program groups its metrics into 6 categories: Area/Edge, Shape, Core Area, Contrast, Aggregation and Diversity at three different scales: patch, class and landscape. In context of our work, the patch scale would refer to individual community polygons, the class scale would refer to all polygons of the same community and the landscape scale would refer to the entire wetland. Since all the study wetlands were visited and visually inspected, it was determined that the individual vegetation community edge did not differ from the center of the vegetation community so no Core Area metrics were generated as those metrics represent differences between the Core Area and the edges of patches. Given the relatively small size of our sites and the lack of discernable edge to each community patch, we did not generate any contrast metrics as they look at patch contrast in terms of edge.

A.3.2.2.5 Discussion

A.3.2.2.5.1 Determining equipment error

The error of the Geneq device was determined through a comparison with the Leica CS15 system. The Leica system was deployed at a field location with the base station placed at a prominent point. The rover unit was then used to measure 59 different points through the area. These points were taken at prominent field land marks (e.g. fence posts) that could be identified in imagery. At each point, the Geneq unit was also used to measure the same point providing us with two measures of each point. The Euclidian distance was calculated between each point in the Geneq layer and the point in the Leica layer that is closest to it using a search radius of 2.5 meters. The search radius was determined using the error specifications for the Geneq device (2.5 meters) knowing that none of the sample points were closer than 2.5 meters. The average distance between the device points was 0.87 meters which provides a Root Mean Square Error (RMSE) of 0.93 and a 2dRMSE of 1.86 meaning that the Geneq had a 95% chance of having the actual point within a 1.86m radius circle around the measured point.

A.3.2.2.5.2 Data and formatting error

The polygons that were created in the field had a number of topology errors that needed to be corrected before any analyses could be carried out. The technician creating the polygons was able to monitor their progress on the Trimble device, however certain topology errors occurred (Figure A.3.2.1). When smaller vegetation communities were positioned entirely inside larger vegetation communities, the polygon for the larger community often totally overlapped the smaller polygon. This meant that the smaller polygon would need to be clipped out from the larger to get an accurate representation of the wetland composition. Once all the topology corrections were made, the vector data could be converted to raster. To minimize error, a cell size of 0.5m was used in the conversion. This size was chosen as it corresponded with the smallest vegetation community mapped. While a smaller cell-size reduces error in vector-to-raster conversion, there is still some error present. Figure A.3.2.2 shows the original vector data over laid on the raster conversion. There is a noticeable difference in the appearance of the

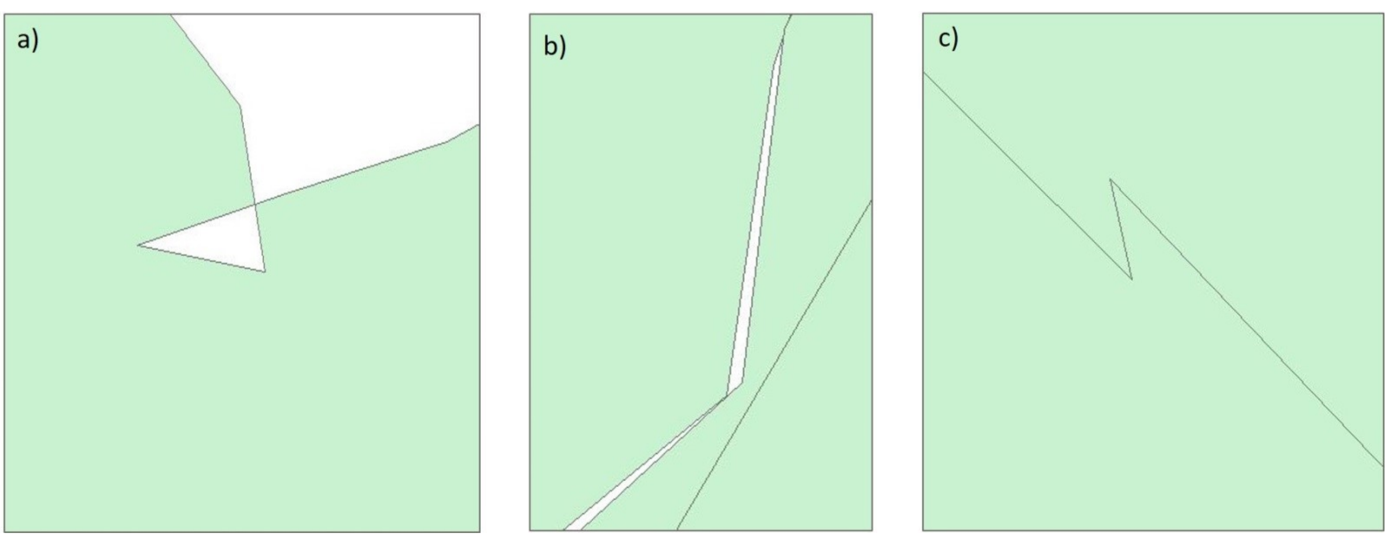

Figure A.3.2.1: Examples of common topology errors observed in polygon creation including a) loops, b) slivers between adjacent polygons and c) switchbacks.

communities, specifically along the edges and the differences in the area and perimeter values can be seen in table A.3.2.1. Since we did not measure the actual area of the vegetation communities while we were in the field, we do not have the known area values from which we could calculate the error created in the vector-to-raster conversion. Instead we can acknowledge that some error is present by visualizing the error seen in the conversion process.

Figure A.3.2.2: Overlay of original vector map on top of the resulting raster conversion. The most noticeable changes being the jagged quality to the edges of the raster map.

Table A.3.2.1 A comparison of the perimeter (m) and area (m²) values of the same map in vector and converted to raster.

Community	Site Code	Perimeter (m)		Area (m²)	
		Vector	Raster	Vector	Raster
gc-cirsarv	cd_67	297	202	598	594
ne-beckman	cd_67	8	10	3	4
ne-careath	cd_67	29	38	28	29
ne-poa	cd_67	15	12	14	14
ts-salixspp	cd_67	128	162	339	340

A.3.2.2.5.3 User error and uncertainty

Human factors contributing to error and uncertainty can be seen as errors in polygon creation, and uncertainty in delineation. It is important to remember that wetlands are not always easy to navigate on foot. Wetlands are often hummocky, plagued by deadfall and are generally difficult to walk through. The easiest user errors to detect are those caused by accidents in the field, such as the tech stumbling and accidentally inserting a point in the map being created Figure A.3.2.3. Errors such as these are easily remedied in the topology correction step. User uncertainty in delineation cannot be remedied, instead it introduces a factor of uncertainty. We can compare the maps made two weeks apart in the 2016 field season to get a better glimpse at the impact of user interpretation on delineation as the vegetation should not have changed much in two weeks. In figure A.3.2.4 we can see that there are noticeable differences between the maps created two weeks apart. Many polygons have similar shape, but the shapes are not exact between both visits. Table A.3.2.2 shows the magnitude of the difference in the area and perimeter values between the two sites. While user interpretation does play a role, we know that natural factors also play a role, despite the relatively short time between visits. For the example shown in figure A.3.2.4, the technician observed that on the second visit the community of "ne-beckman" (*Beckmania syzigachne*) was noticeably diminished and encroached by the dominant "gc-cirsarv" (*Cirsium arvense*). The lack of moisture throughout the summer meant that the *B. syzigachne* community, a wetland obligate species, was shrinking and that difference was noticeable after two weeks. Any other differences, however, were due to interpretation errors on the part of the technician resulting from uncertainty in delineating community boundaries.

A.3.2.2.6 Conclusion

Landscape ecology is an extremely useful tool for landscape management, conservation and restoration, but, as we have seen, error and uncertainty are present at many different levels in the process. The case study presented here highlights some of the common sources of error and uncertainty that occur when using spatial data. Some error, such as device error, is quantifiable while other error sources can only be visualized and not quantified. It is also important to know the characteristics of the landscape being studied and how that will influence error and uncertainty. In our case study, the Prairie Pothole Region has dynamic characteristics, such as a varying hydrological regime, that leads to relatively rapid changes in the measured wetland communities over a short period of time. Knowing this, we can frame our uncertainty against the natural variability that we have observed for this ecosystem to better inform error and uncertainty estimates. Regardless of the ecosystem being examined, awareness of error and uncertainty is important both before analysis, so steps can be taken to reduce error, and after analysis, so the quality of the final results can be interpreted with an understanding of the amount of error and uncertainty involved in the work. This provides a more accurate interpretation of landscape level measurements.

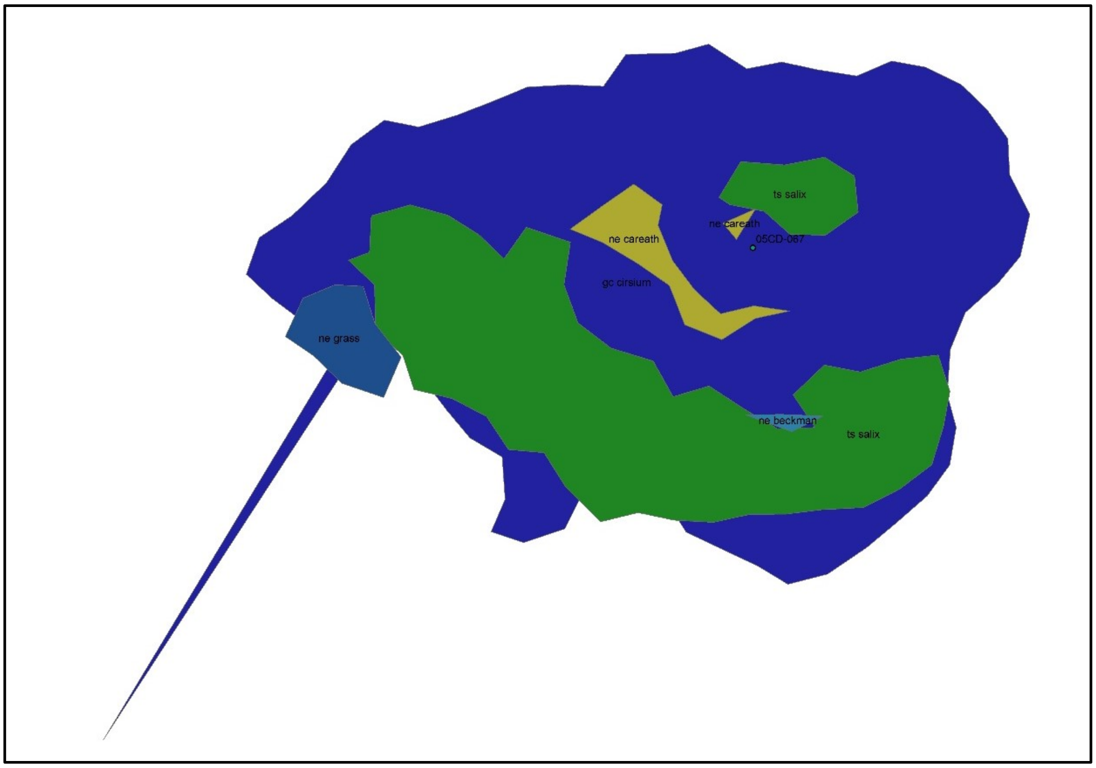

Figure A.3.2.3: An example of a user error when creating the polygons. Accidentally brushing against the touch screen of the Trimble unit can drop a wayward point creating exaggerated errors.

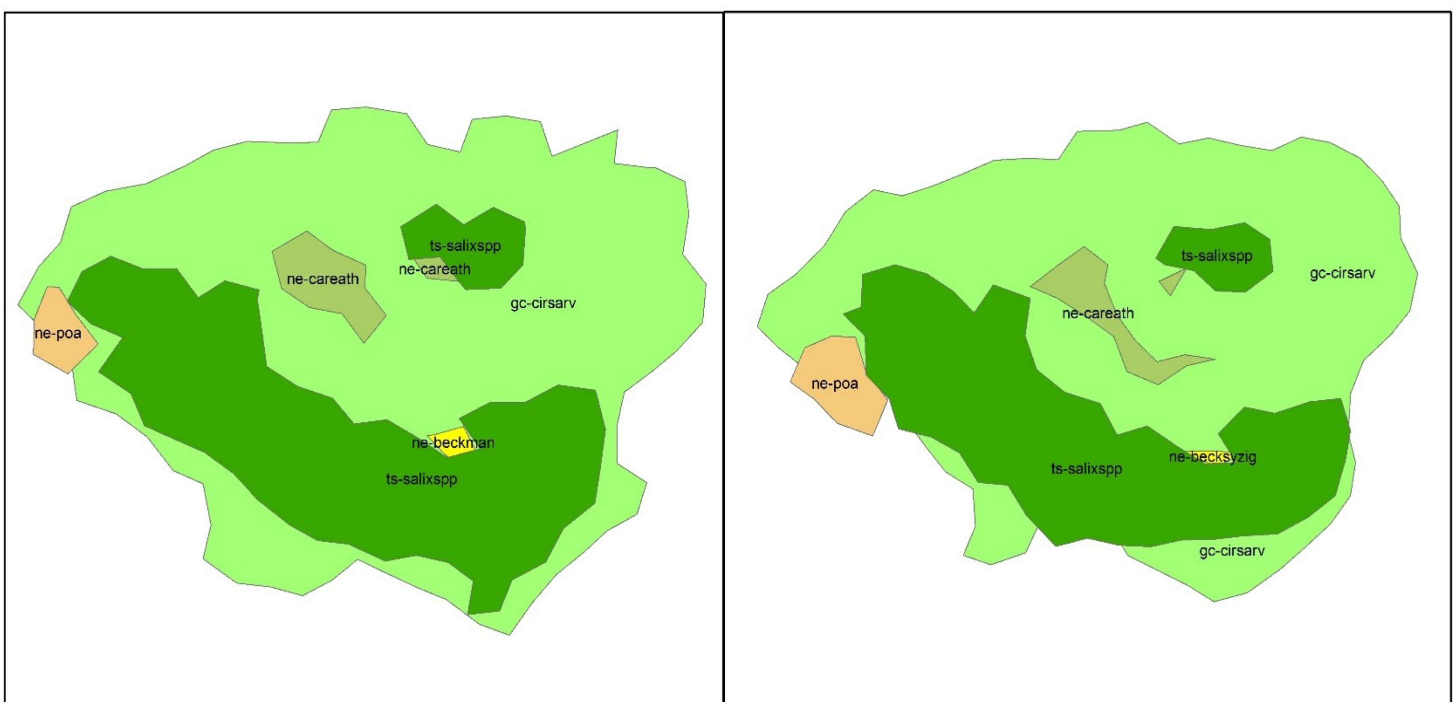

Figure A.3.2.4: A comparison of polygon made for one site mapped two weeks apart in August 2016. The map on the left was created first and the map on the right was created two weeks later.

Table A.3.2.2: A comparison of the perimeter (m) and area (m²) values of maps made of the same site on two separate visits, two weeks apart.

Community	Site Code	Perimeter (m)		Area (m²)	
		Visit 1	Visit 2	Visit 1	Visit 2
gc-cirsarv	cd_67	296.7	267.1	597.8	577.1
ne-beckman	cd_67	7.9	7.2	3.4	1.9
ne-careath	cd_67	29.3	39.5	27.9	26.8
ne-poa	cd_67	14.8	20.4	13.9	25.8
ts-salixspp	cd_67	128.0	120.9	338.9	318.7

Appendix 3.3: List of all wetland assemblage types considered for analysis and their raster identification code.

Table A.3.3.1: All vegetation growth forms from the Ontario Wetland Evaluation System used in vegetation assemblage classification.

Code	Description	Criteria
h	deciduous trees > 6 m tall	This is dominant if > 25% cover
c	coniferous trees > 6 m tall	This is dominant if > 25% cover
dh	dead trees > 6 m tall	This is dominant if > 10% total cover
ts	tall shrubs (1-6 m tall)	This is dominant if > 25% cover
ls	low shrubs (< 1 m tall)	This is dominant if > 25% cover
ds	dead shrubs (< 6m tall)	This is dominant if > 25% cover
re	robust emergents (rushes and cattails)	This is dominant if > 25% cover
be	broad-leaved emergents (*Sagittaria cuneata*, Calla Lily, *Alisma plantago-aquatica*)	This is dominant if > 25% cover
ne	narrow-leaved emergents (sedges and grasses)	This is dominant if > 25% cover
gc	ground cover (herbaceous broad leafed veg)	This is dominant if > 25% cover
f	rooted floating vegetation (water lilies, *Potamogeton natans*, *Polygonum amphibium*)	This is dominant if > 25% cover
ff	free-floating, not rooted in the sediment (*Lemna* spp., *Wolfia* spp., *Ricciocarpus natans*)	This is dominant if > 25% cover
su	submergent (*Ceratophyllum demersum*, *Myriophyllum* spp. most Potamogetons)	This is dominant if > 25% cover
m	moss	This is dominant if > 25% cover
u	unvegetated (sand, mud flat)	This is dominant if > 25% cover

Table A.3.3.2: Cover classes and their corresponding raster ID codes

Cover Class	Assemblage Name	Dominant Cover	Raster ID
Broad-leaved emergent	be-alisma	*Alisma triviale*	1
	be-callpal	*Calla palustris*	2
	be-hippuris	*Hippuris vulgaris*	3
Standing dead	dh-standingdead	*Standing dead*	4
Ground cover	gc-bidens	*Bidens cernua*	5
	gc-chenalb	*Chenopodium album*	6
	gc-equisetum	*Equisetum* spp.	7
	gc-mentha	*Mentha arvensis*	8
	gc-petasites	*Petasites frigidus*	9
	gc-plascoul	*Plagiobothrys scouleri*	10
	gc-polygonum	*Polygonum* spp.	11
	gc-polylap	*Persicaria lapathifolia*	12
	gc-poteanser	*Potentilla anserina*	13
	gc-ranunc	*Ranunculus* spp.	14
	gc-rumintro	Non-native *Rumex* spp.	15
	gc-rumnative	*Native Rumex spp.*	16
	gc-salicornia	*Salicornia rubra*	17
	gc-solidago	*Solidago altissima*	18
	gc-weedy	*Dominant weedy species including: Sonchus spp., Cirsium arvense, Taraxacum officinale*	19
Trees	h-populus	*Populus tremuloides*	20
Moss	m-moss	Moss	21
Narrow-leaved emergent	ne-agrscabr	*Agrostis scabra*	22
	ne-calcan	*Calamagrostis canadensis*	23
	ne-carexspp	*Any Carex* species not given their own cover class	24
	ne-carsychn	*Carex sychnocephala*	25
	ne-descesp	*Deschampsia cespitosa*	26
	ne-eleoacic	*Eleocharis acicularis*	27
	ne-eleoch	*Eleocharis* spp. not *E. acicularis*	28
	ne-elyrepen	*Elymus repens*	29
	ne-elytrach	*Elymus trachycaulus*	30

Cover Class	Assemblage Name	Dominant Cover	Raster ID
Narrow-leaved emergent	ne-grassspp	Any grass species not given their own cover class	31
	ne-hordjub	*Hordeum jubatum*	32
	ne-juncbal	*Juncus balticus*	33
	ne-juncus	Any *Juncus* species not *J. balticus*	34
	ne-oblcarex	Obligate *Carex* species including: *C. atherodes, C. pelli, C. retrosoa, C. utriculata*	35
	ne-oblgrass	Obligate grass species including: *Beckmania syzigachne, Alopecurus aequalis, Scolochloa festucacea*	36
	ne-phalaru	*Phalarius arundinacea*	37
	ne-phleum	*Phleum pratense*	38
	ne-poa	*Poa spp.*	39
	ne-poapal	*Poa palustris*	40
	ne-poaprate	*Poa pratensis*	41
	ne-scirpun	*Schoenoplectus pungens*	42
	ne-sparaganium	*Sparganium spp.*	43
Robust emergent	re-scirpus	Any bullrush species not given its own cover class	44
	re-scirpval	*Schoenoplectus tabernaemontani*	45
	re-typha	*Typha latifolia*	46
Tall shrubs	ts-salix	*Salix* spp.	47
Unvegetated	u-aquamoss	Aquatic moss	48
	u-bareground	Bare ground	49
	u-crop	Agricultural cultivars	50
	u-drawdown	Exposed, saturated sediment	51
	u-openwater	Open water	52

Appendix 3.4: Edge contrasts based on vegetation growth form.

Edge contrasts are presented here as a matrix of comparisons based on growth-forms. The growth forms were created based on the observed average difference in heights between cover classes with the intent that edge difference was of primary importance for fauna within the wetland. Smaller values indicate a similarity between the habitat height while larger values indicate a difference in height.

	Broad-leaved emergent	Standing dead	Ground-cover	Trees	Moss	Narrow-leaved emergent	Robust emergent	Shrubs	Unvegetated
Broad-leaved emergent	0	0.7	0.2	0.8	0.2	0.4	0.6	0.7	0.3
Standing dead	0.7	0	0.4	0.1	0.7	0.3	0.2	0.1	0.8
Ground-cover	0.2	0.4	0	0.4	0.2	0.1	0.3	0.5	0.3
Trees	0.8	0.1	0.4	0	0.9	0.6	0.4	0.2	1
Moss	0.2	0.7	0.2	0.9	0	0.3	0.4	0.7	0.1
Narrow-leaved emergent	0.4	0.3	0.1	0.6	0.3	0	0.2	0.3	0.4
Robust emergent	0.6	0.2	0.3	0.4	0.4	0.2	0	0.2	0.6
Shrubs	0.7	0.1	0.5	0.2	0.7	0.3	0.2	0	0.8
Unvegetated	0.3	0.8	0.3	1	0.1	0.4	0.6	0.8	0

Appendix 3.5: Fragstats metric choices for class and landscape level

Table A.3.5.1: FRAGSTATS class metrics chosen for spatial MMI creation

Class Metrics n=23			
Metric Type	**Metric**	**Code**	**Description**
Area/Edge	Total Area	CA/TA	Sum of the areas of all patches of the same class
Area/Edge	Percentage of the Landscape	PLAND	Sum of the areas of all patches of the same class, divided by the total landscape area (proportional abundance)
Area/Edge	Largest Patch Index	LPI	Percentage of the landscape covered by the largest patch
Area/Edge	Total Edge	TE	Sum of the lengths of all edge segments for each patch type
Area/Edge	Edge Density	ED	Sum of the lengths of all edge segments for each patch type divided by the total landscape area
Core Area	Total Core Area	TCA	Sum of core areas of each patch of patch type
Core Area	Core Area Percentage of Landscape	CPLAND	Sum of core areas for each patch divided by the total landscape area
Core Area	Number of Disjunct Core Areas	NDCA	Number of disjunct core areas contained within each patch of the corresponding type
Core Area	Disjunct Core Area Density	DCAD	Sum of the disjunct core areas contained within each patch type divided by total landscape area
Contrast	Contrast-Weighted Edge Density	CWED	Sum of the lengths of each edge segment per patch type, multiplied by the corresponding contrast weight divided by total landscape area
Contrast	Total Edge Contrast Index	TECI	Sum of the lengths of each edge segment per patch type, multiplied by the corresponding contrast weight divided the lengths of all edge edges of the same type
Aggregation	Interspersion and Juxtaposition Index	IJI	Sum of the length of each unique edge type involving the corresponding patch type divided by the total length of edge involving the same type, multiplied by the logarithm of the same quantity summed over each unique edge type divided by the logarithm of the number of patch types minus 1
Aggregation	Percentage of Like Adjacencies	PLADJ	Number of like adjacencies involving the focal class divided by the total number of cell adjacencies involving the focal class

Class Metrics n=23			
Metric Type	**Metric**	**Code**	**Description**
Aggregation	Aggregation Index	AI	Number of like adjacencies involving the corresponding class, divided by the maximum possible number of like adjacencies involving the corresponding class.
Aggregation	Clumpiness Index	CLUMPY	Proportional deviation of the proportion of like adjacencies involving the corresponding class from that expected under a spatially random distribution.
Aggregation	Landscape Shape Index	LSI	0.25 the sum of the entire landscape boundary and all edge segments within the landscape boundary involving the corresponding patch type divided by the square root of the total landscape area
Aggregation	Normalized Landscape Shape Index	NLSI	Total length of edge of the corresponding class given in number of cell surfaces minus the minimum length of class edge possible for a maximally aggregated class
Aggregation	Patch Cohesion Index	COHESION	1 minus the sum of patch perimeter divided by the sum of patch perimeter times the square root of patch area divided by 1 minus 1 over the square root of the total number of cells in the landscape
Aggregation	Number of Patches	NP	Number of patches of the corresponding type
Aggregation	Patch Density	PD	Number of patches of the corresponding type divided by the total landscape area
Aggregation	Landscape Division Index	DIVISION	1 minus the sum of patch area divided by total landscape area, quantity squared, summed across all patches of the corresponding patch type
Aggregation	Splitting Index	SPLIT	Total landscape area squared divided by the sum of patch area squared summed across all patches of all the corresponding patch type
Aggregation	Effective Mesh Size	MESH	Sum of patch area squared, summed across all patches of the corresponding patch type, divided by total landscape area

Table A.3.5.2: FRAGSTATS landscape metrics chosen for spatial MMI creation.

Landscape Metrics n=29			
Metric Type	**Metric**	**Code**	**Description**
Area/Edge	Total Area	TA	Total area of the landscape
Area/Edge	Largest Patch Index	LPI	Area of the largest patch in the landscape divided by the total landscape area
Area/Edge	Total Edge	TE	Sum of the lengths of all edge segments in the landscape
Area/Edge	Edge Density	ED	Sum of the lengths of all edge segments in the landscape divided by the total landscape area
Core Area	Total Core Area	TCA	Sum of core area of each patch
Core Area	Number of Disjunct Core Areas	NDCA	Sum of the number of disjunct core areas contained within each patch in the landscape
Core Area	Disjunct Core Area Density	DCAD	Sum of the number of disjunct core areas contained within each patch in the landscape divided by total landscape area
Contrast	Contrast-Weighted Edge Density	CWED	Sum of the lengths of each edge segment in the landscape multiplied by the contrast weight divided by the total landscape area
Contrast	Total Edge Contrast Index	TECI	Sum of the lengths of each edge segment in the landscape multiplied by the contrast weight divided by the total length of edge in the landscape
Aggregation	Contagion Index	CONTAG	Minus the sum of the proportional abundance of each patch type multiplied by the proportion of adjacencies between cells of that patch type and another patch type, multiplied by the logarithm of the same quantity summed over each unique adjacency type and each patch type, divided by 2 times the logarithm of the number of patch types.
Aggregation	Interspersion and Juxtaposition Index	IJI	Sum of the length of each unique edge type divided by the total landscape edge, multiplied by the logarithm of the same quantity summed over each unique edge the divided by the logarithm of the number of patch types minus 1, divided by 2
Aggregation	Percentage of Like Adjacencies	PLADJ	Sum of the number of like adjacencies for each patch type, divided by the total number of cell adjacencies in the landscape

Landscape Metrics n=29			
Metric Type	**Metric**	**Code**	**Description**
Aggregation	Aggregation Index	AI	Number of like adjacencies involving the corresponding class, divided by the max possible number of like adjacencies involving the corresponding class, summed over all classes
Aggregation	Landscape Shape Index	LSI	0.25 times the sum of the entire landscape boundary and all edge segments within the landscape boundary divided by the square root of the total landscape area.
Aggregation	Patch Cohesion Index	COHESION	1 minus the sum of patch perimeter divided by the sum of patch perimeter times the square root of patch area for all patches in the landscape divided by 1 minus 1 over the square root of the total number of cells in the landscape
Aggregation	Number of Patches	NP	Equals the number of patches in the landscape
Aggregation	Patch Density	PD	Number of patches in the landscape divided by total landscape area
Aggregation	Landscape Division Index	DIVISION	1 minus the sum of patch area divided by total landscape area, quantity squared, summed across all patches in the landscape
Aggregation	Splitting Index	SPLIT	total landscape area squared, divided by the sum of patch area squared, summed across all patches in the landscape
Aggregation	Effective Mesh Size	MESH	1 divided by total landscape area, multiplied by the sum patch area squared, summed across all patches in the landscape
Diversity	Patch Richness	PR	Number of different patch types present in the landscape
Diversity	Patch Richness Density	PRD	Number of different patch types present in the landscape divided by total landscape area
Diversity	Relative Patch Richness	RPR	Number of different patch types present within the landscape boundary divided by the maximum potential number of patch types specified by the user
Diversity	Shannon's Diversity Index	SHDI	minus the sum across all patch types of the proportional abundance of each patch type multiplied by that proportion
Diversity	Simpson's Diversity Index	SIDI	1 minus the sum, across all patches, of the proportional abundance of each patch type squared

Landscape Metrics n=29			
Metric Type	**Metric**	**Code**	**Description**
Diversity	Modified Simpson's Diversity Index	MSIDI	minus the log of the sum, across all patch types, of the proportional abundance of each patch type squared
Diversity	Shannon's Evenness Index	SHEI	minus the sum, across all patch types, of the proportional abundance of each patch type multiplied by that proportion divided by the log of the number of patch types.
Diversity	Simpson's Evenness Index	SIEI	1 minus the sum, across all patches, of the proportional abundance of each patch type squared, divided by 1 minus the number of patch types
Diversity	Modified Simpson's Evenness Index	MSIEI	minus the log of the sum, across all patch types, of the proportional abundance of each patch type squared, divided by the log of the number of patch types

Appendix 3.6: All metrics with sufficient range (i.e., |95th percentile – 5th percentile| > 0) that differed significantly between low and high disturbance sites.

Metric	U-statistic	p-value
ne.oblcarex_DCAD	37.5	0.0065
ne.oblcarex_NLSI	37.5	0.0065
ne.oblcarex_NP	38.5	0.0077
ne.oblcarex_NDCA	38.5	0.0077
ne.oblcarex_PD	38.5	0.0077
ne.oblcarex_LSI	39.5	0.0092
LSI_Land	37.0	0.0140
ne.oblcarex_ED	42.5	0.0151
ts.salix_NP	52.0	0.0170
ts.salix_CA	52.0	0.0171
ts.salix_PLAND	52.0	0.0171
ts.salix_PD	52.0	0.0171
ts.salix_LPI	52.0	0.0171
ts.salix_TE	52.0	0.0171
ts.salix_ED	52.0	0.0171
ts.salix_LSI	52.0	0.0171
ts.salix_TCA	52.0	0.0171
ts.salix_CPLAND	52.0	0.0171
ts.salix_NDCA	52.0	0.0171
ts.salix_DCAD	52.0	0.0171
ts.salix_CWED	52.0	0.0171
ts.salix_TECI	52.0	0.0171
ts.salix_CLUMPY	52.0	0.0171
ts.salix_PLADJ	52.0	0.0171
ts.salix_IJI	52.0	0.0171
ts.salix_COHESION	52.0	0.0171
ts.salix_DIVISION	52.0	0.0171
ts.salix_MESH	52.0	0.0171
ts.salix_SPLIT	52.0	0.0171
ts.salix_AI	52.0	0.0171
ts.salix_NLSI	52.0	0.0171
ne.oblcarex_DIVISION	43.5	0.0177
ne.oblcarex_TE	43.5	0.0178
ne.oblcarex_SPLIT	43.5	0.0178
ne.oblcarex_CA	45.5	0.0242
ne.oblcarex_TCA	45.5	0.0242
ne.oblcarex_MESH	47.5	0.0325
ne.oblcarex_PLAND	47.5	0.0326
ne.oblcarex_LPI	47.5	0.0326

Metric	U-statistic	p-value
ne.oblcarex_CPLAND	47.5	0.0326
ne.oblcarex_IJI	48.5	0.0377
ne.oblcarex_PLADJ	49.5	0.0434
ne.oblcarex_COHESION	49.5	0.0434
ne.oblcarex_CWED	51.0	0.0463
NDCA_Land	45.5	0.0480
AI_Land	122.0	0.0568
ne.oblcarex_TECI	53.0	0.0612
ne.oblcarex_CLUMPY	53.5	0.0742
ne.oblcarex_AI	53.5	0.0742
TE_Land	50.0	0.0811
be.alisma_CA	97.5	0.1655
be.alisma_PLAND	97.5	0.1655
be.alisma_NP	97.5	0.1655
be.alisma_PD	97.5	0.1655
be.alisma_LPI	97.5	0.1655
be.alisma_TE	97.5	0.1655
be.alisma_ED	97.5	0.1655
be.alisma_LSI	97.5	0.1655
be.alisma_TCA	97.5	0.1655
be.alisma_CPLAND	97.5	0.1655
be.alisma_NDCA	97.5	0.1655
be.alisma_DCAD	97.5	0.1655
be.alisma_CWED	97.5	0.1655
be.alisma_TECI	97.5	0.1655
be.alisma_CLUMPY	97.5	0.1655
be.alisma_PLADJ	97.5	0.1655
be.alisma_IJI	97.5	0.1655
be.alisma_COHESION	97.5	0.1655
be.alisma_DIVISION	97.5	0.1655
be.alisma_MESH	97.5	0.1655
be.alisma_SPLIT	97.5	0.1655
be.alisma_AI	97.5	0.1655
be.alisma_NLSI	97.5	0.1655
ED_Land	57.0	0.1690

Appendix 3.7: Fifth and ninety-fifth percentiles for all significant metrics. Note that the range test required the calculation of the 5th and 95th percentiles for each metric.

Metric	5th Percentile	95th Percentile
be.alisma_CA	0.0	0.0
be.alisma_PLAND	0.0	2.1
be.alisma_NP	0.0	0.3
be.alisma_PD	0.0	11.8
be.alisma_LPI	0.0	2.1
be.alisma_TE	0.0	26.9
be.alisma_ED	0.0	78.4
be.alisma_LSI	0.0	0.5
be.alisma_TCA	0.0	0.0
be.alisma_CPLAND	0.0	1.3
be.alisma_NDCA	0.0	0.3
be.alisma_DCAD	0.0	101.8
be.alisma_CWED	0.0	49.4
be.alisma_TECI	0.0	10.1
be.alisma_CLUMPY	0.0	0.3
be.alisma_PLADJ	0.0	33.4
be.alisma_IJI	0.0	7.3
be.alisma_COHESION	0.0	34.1
be.alisma_DIVISION	0.0	0.3
be.alisma_MESH	0.0	0.0
be.alisma_SPLIT	0.0	2.5
be.alisma_AI	0.0	33.8
be.alisma_NLSI	0.0	0.0
ne.oblcarex_CA	0.0	0.7
ne.oblcarex_PLAND	0.0	98.1
ne.oblcarex_NP	0.0	11.7
ne.oblcarex_PD	0.0	2268.0
ne.oblcarex_LPI	0.0	98.1
ne.oblcarex_TE	0.0	1582.9
ne.oblcarex_ED	0.0	2301.9
ne.oblcarex_LSI	0.0	6.4
ne.oblcarex_TCA	0.0	0.5
ne.oblcarex_CPLAND	0.0	82.2
ne.oblcarex_NDCA	0.0	18.8
ne.oblcarex_DCAD	0.0	3049.4
ne.oblcarex_CWED	0.0	689.9
ne.oblcarex_TECI	0.0	40.0
ne.oblcarex_CLUMPY	0.0	1.0
ne.oblcarex_PLADJ	0.0	98.7

Metric	5th Percentile	95th Percentile
ne.oblcarex_IJI	0.0	95.3
ne.oblcarex_COHESION	0.0	100.0
ne.oblcarex_DIVISION	0.0	1.0
ne.oblcarex_MESH	0.0	0.5
ne.oblcarex_SPLIT	0.0	7190.3
ne.oblcarex_AI	0.0	99.6
ne.oblcarex_NLSI	0.0	0.5
ts.salix_CA	0.0	0.2
ts.salix_PLAND	0.0	57.6
ts.salix_NP	0.0	8.7
ts.salix_PD	0.0	3131.6
ts.salix_LPI	0.0	36.4
ts.salix_TE	0.0	773.7
ts.salix_ED	0.0	2185.2
ts.salix_LSI	0.0	4.9
ts.salix_TCA	0.0	0.2
ts.salix_CPLAND	0.0	29.4
ts.salix_NDCA	0.0	11.7
ts.salix_DCAD	0.0	3687.0
ts.salix_CWED	0.0	1136.7
ts.salix_TECI	0.0	61.5
ts.salix_CLUMPY	0.0	1.0
ts.salix_PLADJ	0.0	96.2
ts.salix_IJI	0.0	77.5
ts.salix_COHESION	0.0	99.1
ts.salix_DIVISION	0.0	1.0
ts.salix_MESH	0.0	0.1
ts.salix_SPLIT	0.0	5434.4
ts.salix_AI	0.0	97.7
ts.salix_NLSI	0.0	0.1
TE_Land	176.9	3272.8
ED_Land	918.6	4666.7
LSI_Land	1.3	6.5
NDCA_Land	1.0	43.4
AI_Land	93.0	99.8

Appendix 3.8: Relationship with top spatial metrics and wetland area.

Table 3.8-1:Metrics from the best 4-metric spatial MMI and their relationship to wetland area.

Metric	Relationship with Disturbance	Metric Group	Relationship with Area
Total edge of obligate *Carex* spp.	Negative	Area/Edge	Some constraint of low values to smaller wetlands.
SPLIT of obligate *Carex* spp.	Negative	Aggregation	No discernable relationship with area.
Edge density of *Salix* spp.	Negative	Area/Edge	Some constraint of high values to smaller wetlands.
Aggregation index of the wetland	Positive	Aggregation	No discernable relationship with area.

Table 3.8-2: Metrics from the best 6-metric spatial MMI and their relationship to wetland area.

Metric	Relationship with Disturbance	Metric Group	Relationship with Area
Percentage of the landscape occupied by obligate *Carex*	Negative	Area/Edge	No discernable relationship with area.
SPLIT of obligate *Carex* spp.	Negative	Aggregation	No discernable relationship with area.
Landscape shape index for *Alisma triviale*	Positive	Aggregation	No discernable relationship with area.
Contrast-weighted edge density of *Salix* spp.	Negative	Contrast	Some constraint of high values to smaller wetlands.
Total edge of the wetland	Negative	Area/Edge	Strong relationship with area.
Aggregation index of the wetland	Positive	Aggregation	No discernable relationship with area.

Table 3.8-3: Metrics from the best 8-metric spatial MMI and their relationship with area.

Metric	**Relationship with Disturbance**	**Metric Group**	**Relationship with Area**
Total edge contrast for obligate *Carex*	Negative	Contrast	No discernable relationship with area.
SPLIT of obligate *Carex* spp.	Negative	Aggregation	No discernable relationship with area.
Division index for *Alisma triviale*	Positive	Aggregation	No discernable relationship with area.
Disjunct core area density of obligate *Carex*	Negative	Core Area	Some constraint of high values to smaller wetlands.
Effective mesh size of obligate *Carex*	Negative	Aggregation	Some constraint of high values to smaller wetlands.
Landscape shape index of *Salix* spp.	Negative	Aggregation	Some constraint of high values to smaller wetlands.
Landscape shape index of the wetland	Negative	Aggregation	No discernable relationship with area.
Aggregation index of the wetland	Positive	Aggregation	No discernable relationship with area.

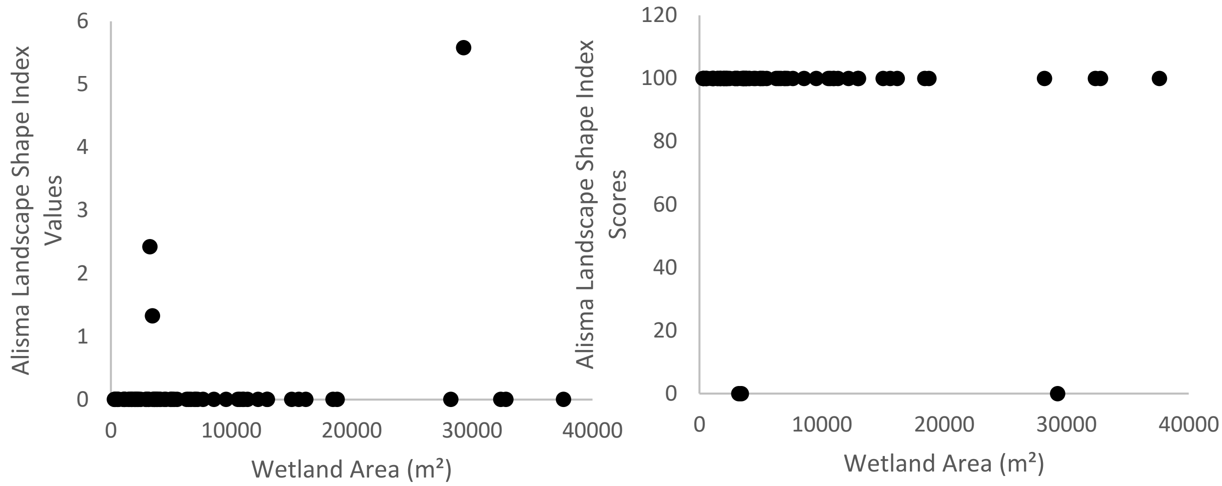

Figure 14: Plots of *Alisma triviale* landscape shape index metric values (left) and metric scores (right) versus wetland area in meters squared.

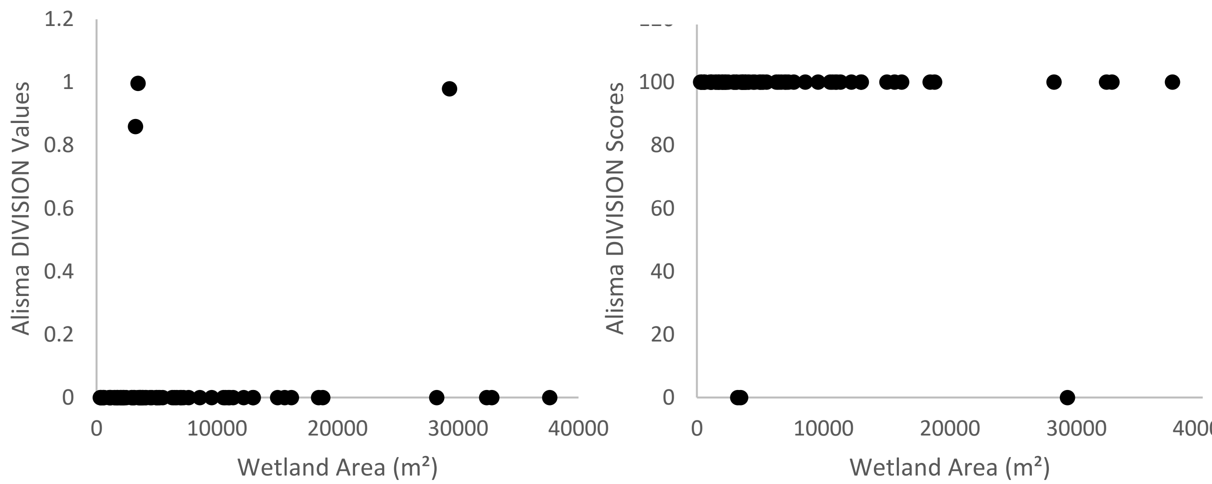

Figure 15: Plots of Alisma triviale division metric values (left) and metric scores (right) versus wetland area in meters squared.

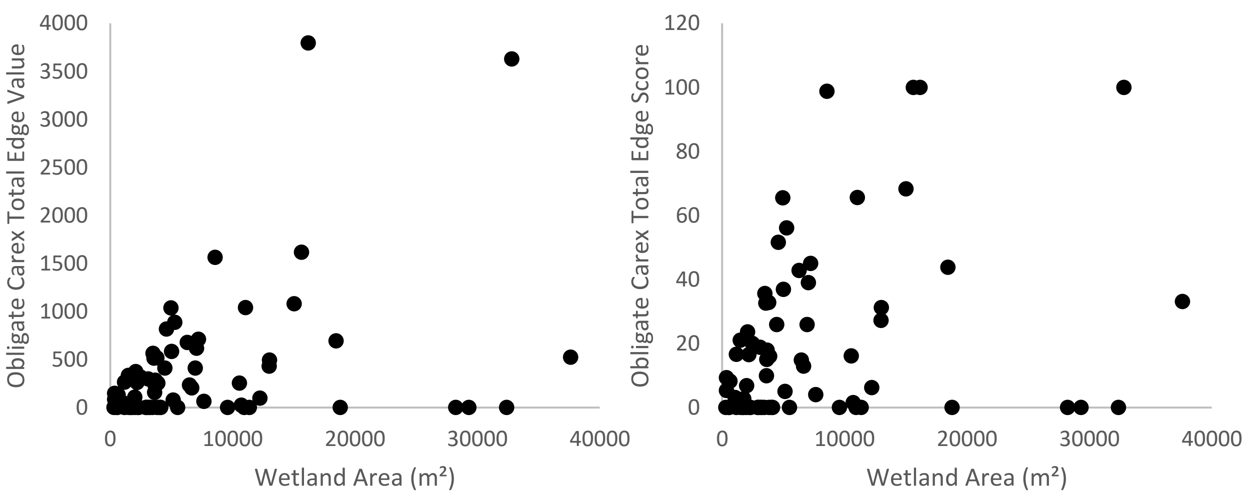

Figure 3: Plots of obligate *Carex* spp. total edge metric values (left) and metric scores (right) versus wetland area in meters squared.

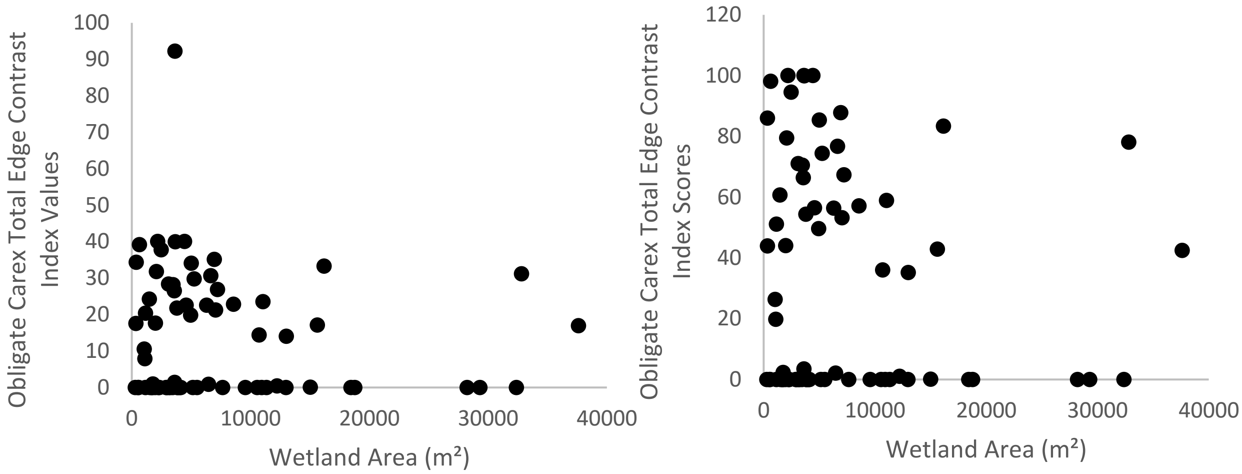

Figure 4: Plots of obligate *Carex* spp. total edge contrast index metric values (left) and metric scores (right) versus wetland area in meters squared.

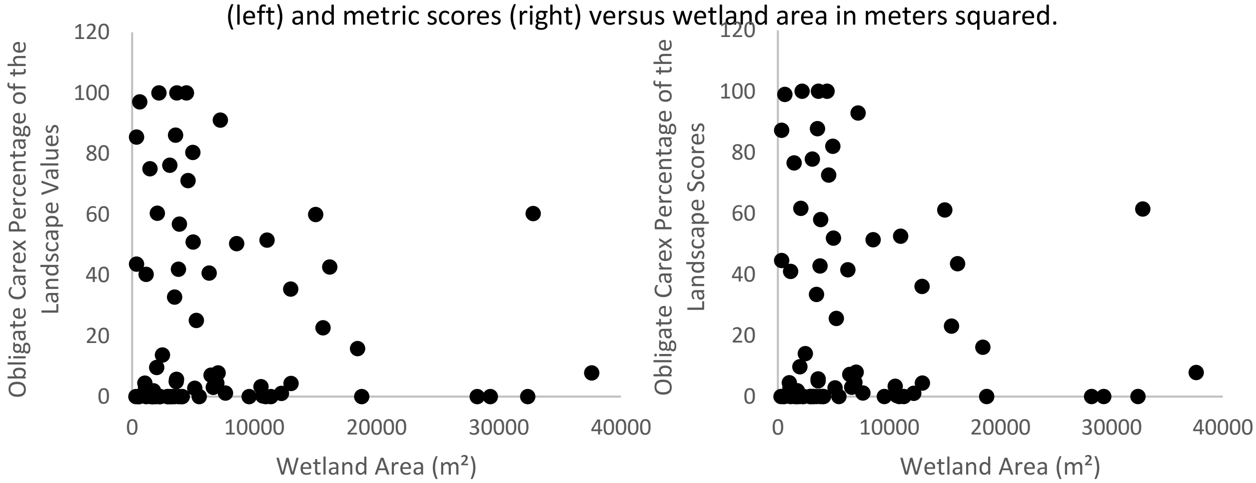

Figure 5: Plots of obligate *Carex* spp. percentage of the landscape metric values (left) and metric scores (right) versus wetland area in meters squared.

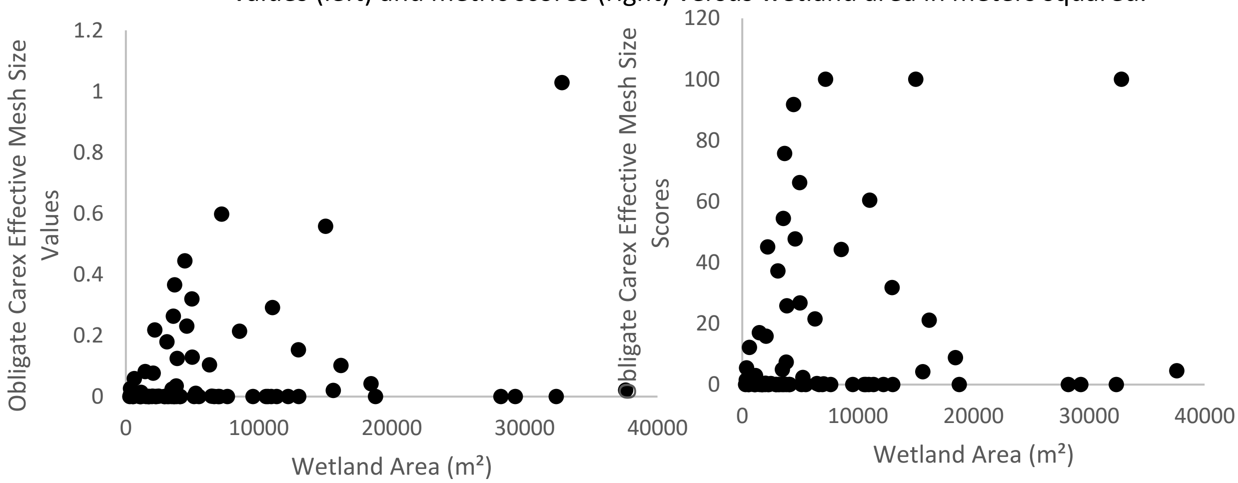

Figure 6: Plots of obligate *Carex* spp. effective mesh size metric values (left) and metric scores (right) versus wetland area in meters squared.

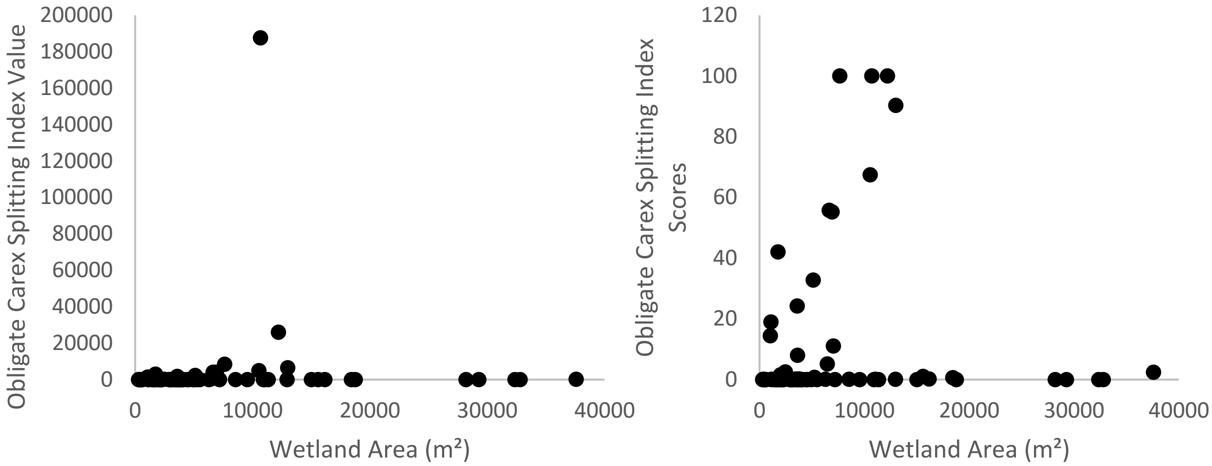

Figure 7: Plots of obligate *Carex* spp. splitting index metric values (left) and metric scores (right) versus wetland area in meters squared.

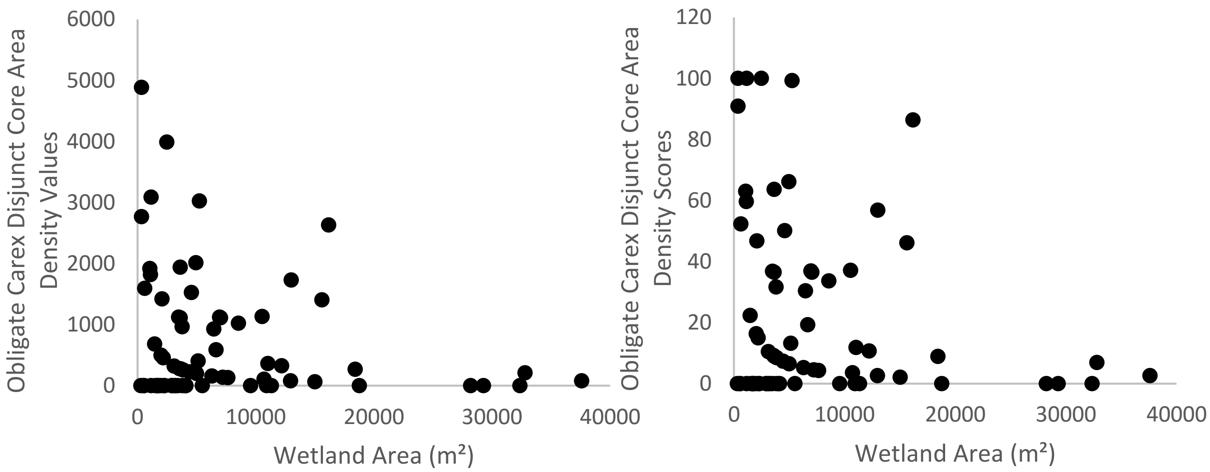

Figure 8: Plots of obligate *Carex* spp. disjunct core area density metric values (left) and metric scores (right) versus wetland area in meters squared.

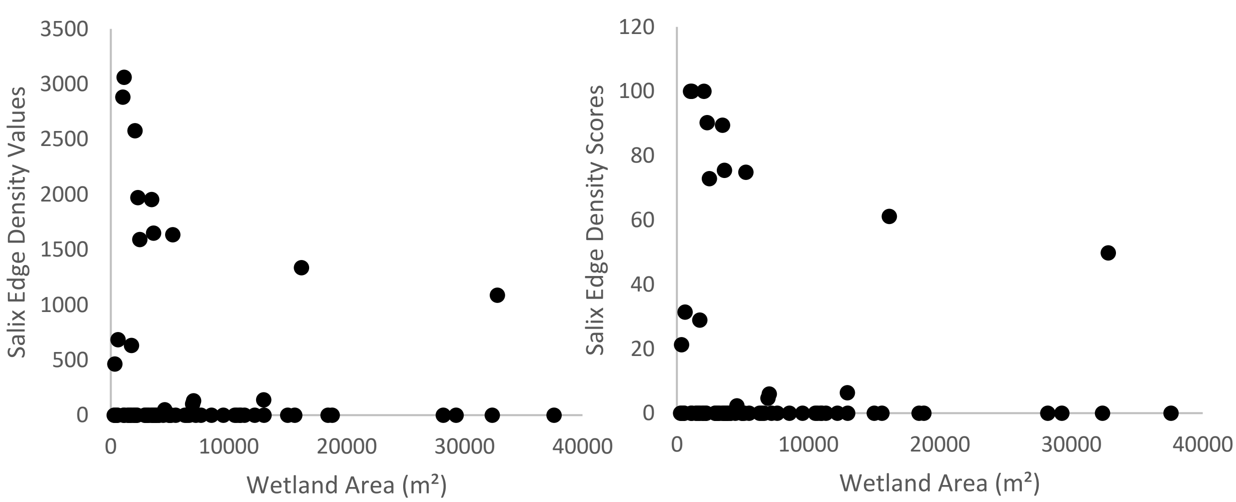

Figure 9: Plots of *Salix* spp. edge density metric values (left) and metric scores (right) versus wetland area in meters squared.

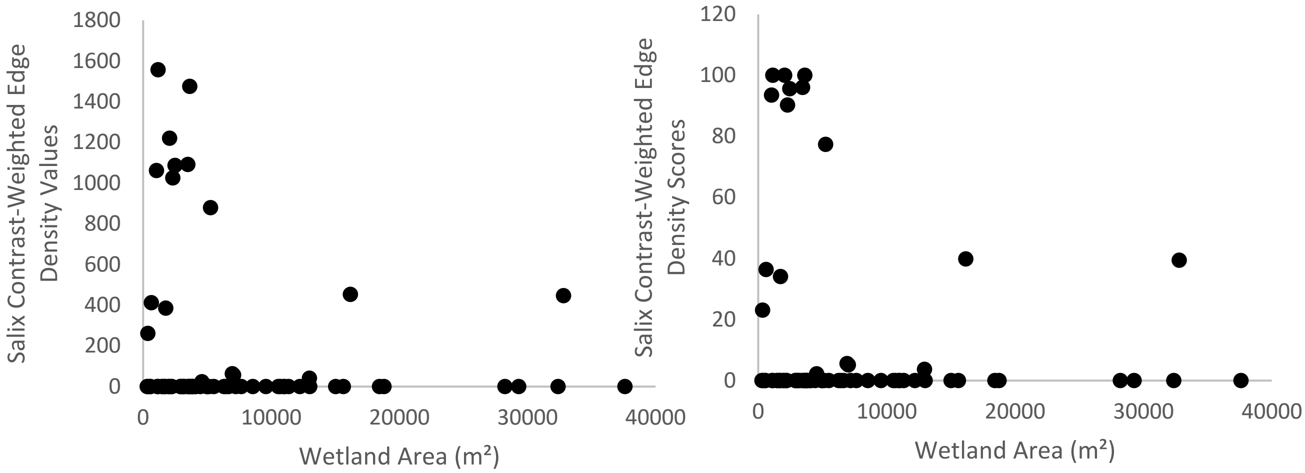

Figure 10: Plots of *Salix* spp. contrast-weighted edge density metric values (left) and metric scores (right) versus wetland area in meters squared.

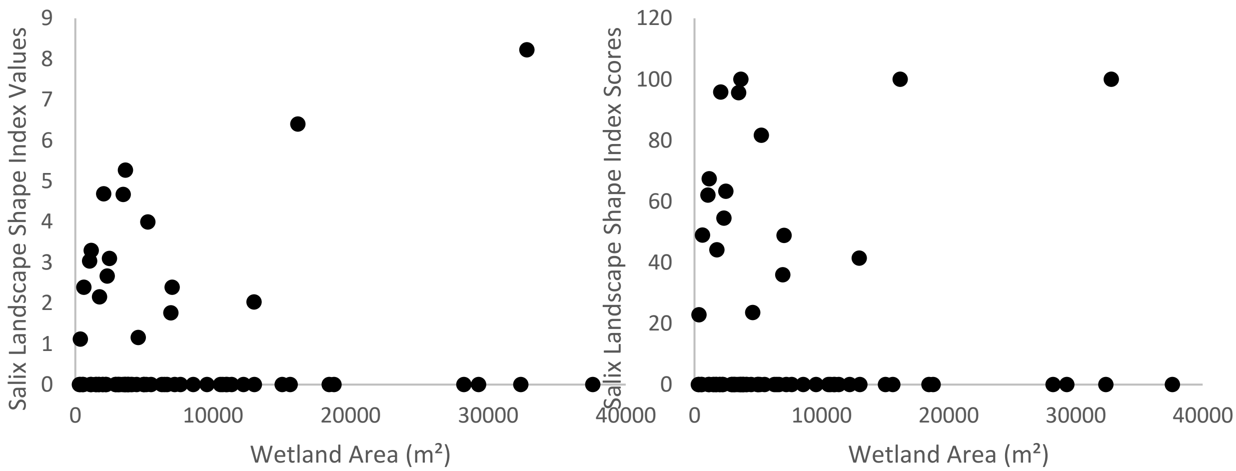

Figure 11: Plots of *Salix* spp. landscape shape index metric values (left) and metric scores (right) versus wetland area in meters squared.

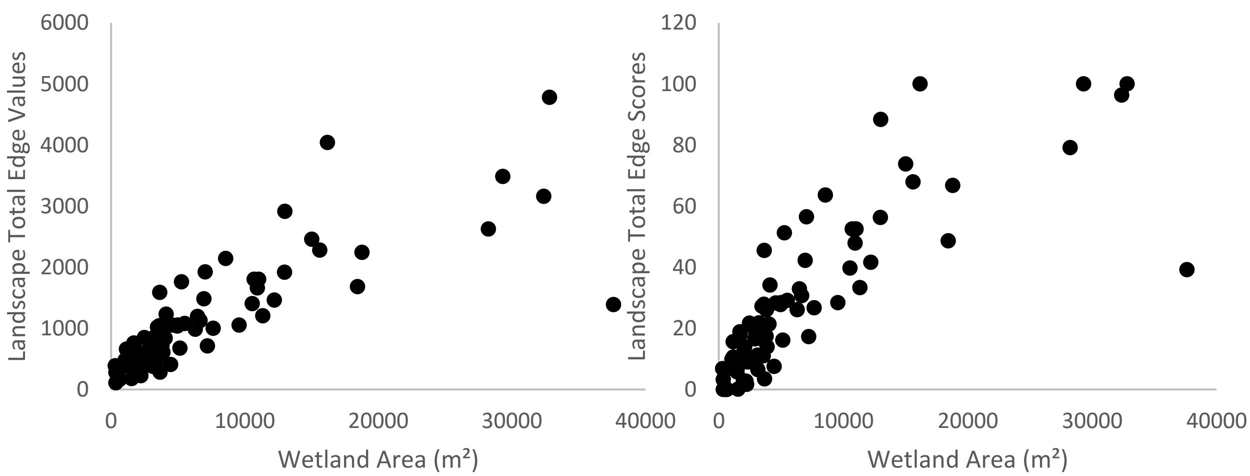

Figure 12: Plots of total edge for the entire wetland metric values (left) and metric scores (right) versus wetland area in meters squared.

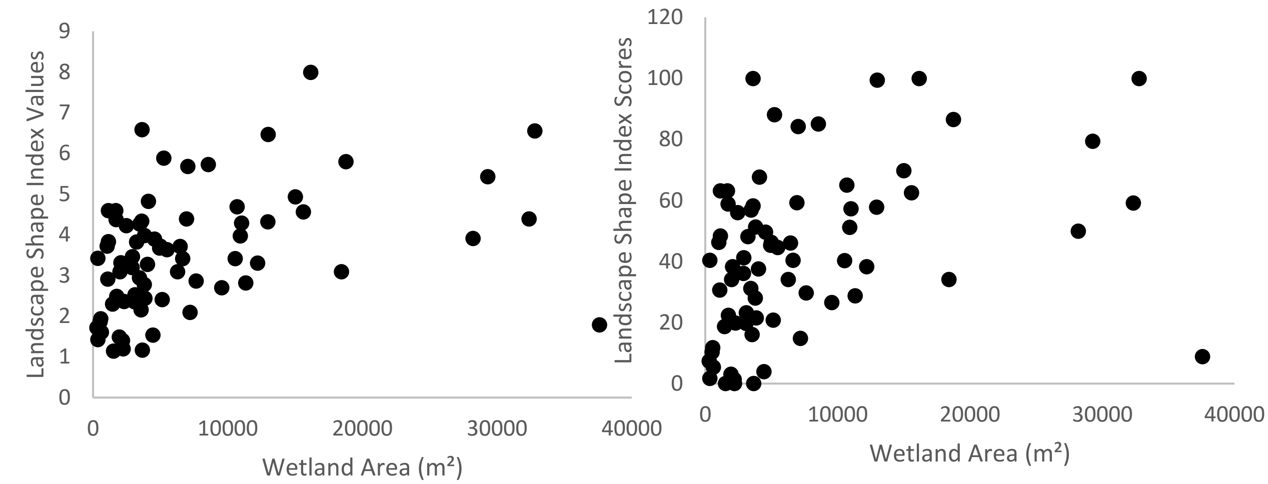

Figure 13: Plots of landscape shape index for the entire wetland metric values (left) and metric scores (right) versus wetland area in meters squared.

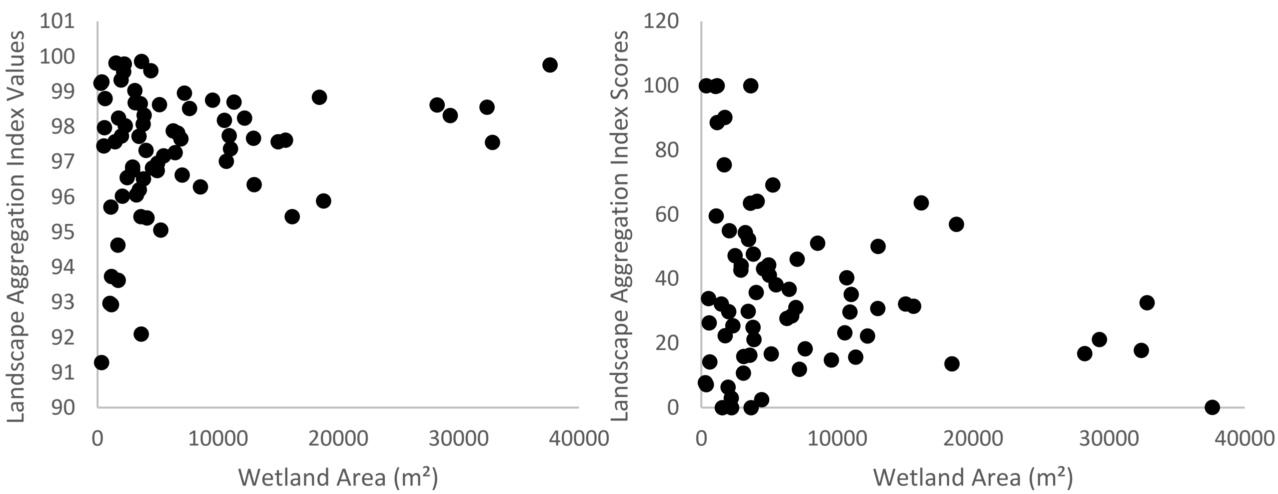

Figure 14: Plots of aggregation index for the entire wetland metric values (left) and metric scores (right) versus wetland area in meters squared.