Quantifying Variation in Wetland Composition and Configuration for Landscape-Scale Reclamation Planning

by

Ian Evans

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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. I understand that my thesis may be made electronically available to the public.

Abstract

Widespread loss and degradation of wetland ecosystems resulting from human land use highlights the need for a reclamation strategy (i.e converting developed land back to its original state) that can sustain ecosystem services provided by wetlands. Successful reclamation planning requires an understanding of the linkages between individual wetlands and the structure of their surrounding landscapes. A parsimonious set of representative metrics, measuring the composition and configuration of wetland landscapes in southern Alberta, was identified using variable reduction procedures, and then related to anthropogenic disturbance with the intent of establishing a continuum of reference conditions for structure of landscapes at varying disturbance levels. The spatial configuration of low-disturbance and high-disturbance landscapes were significantly different from other landscapes, suggesting that a reference condition approach would be appropriate for landscape-scale reclamation. Aggregation metrics quantifying the connectivity, proximity, isolation, contagion, and interspersion of wetland patches were the most commonly identified measures of wetland configuration independent of wetland-proportion in the landscape. Metric values differed significantly between Natural Regions, indicating that reference conditions will likely vary depending on spatial location. Selection and values of representative metrics is impacted by data quality. A framework for wetland reclamation is proposed with the caveat that future research will first need to assess the relationships between landscape characteristics and site-level topography and biophysical conditions.

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Chapter 1 – Introduction

1 – Defining Wetlands

Wetlands are unique ecosystems that occur at the intersection of terrestrial and aquatic ecosystems (Marton et al., 2015). The relationship between the terrestrial and aquatic processes is highly variable in wetland ecosystems, which makes wetlands challenging to define and study. The term 'wetland' is in fact a relatively recent addition to scientific literature which only came into common usage during the second half of the 20th century (National Research Council, 1995). A simple all-encompassing definition of a wetland is an area that is at least periodically inundated with water, resulting in anaerobic soils and vegetation communities that are specially adapted to these conditions (Keddy, 2010; National Research Council, 1995). This broad definition leaves a great deal of room for variation between different types of wetlands.

To better understand and compare research among different types of wetlands, classification systems have been devised to codify different wetland types. However, there is currently no consensus on a universal classification, so the definitions of wetland types found in scientific literature can have inconsistencies. Geography plays an important role in the definition of wetlands. Wetlands can be found all around the world in diverse conditions ranging from the cold subarctic to the warm tropics. It is therefore common for classification schemes to be tailored to a specific geographic region, which can result in a scheme that is not universally applicable (Keddy, 2010). Additionally, language differences can result in different terminology being applied to similar wetland types.

Despite regional differences, wetlands can be broadly categorized as organic (peatland) or mineral, and subdivided into five wetland classes in terms of hydrology and nutrient supply (Keddy, 2010; Figure 1.1). Hydrology and nutrient supply influence the type of vegetation that grows in wetlands, which is a common descriptor for wetland areas.

Organic wetlands (i.e. bogs and fens) are characterized primarily by the accumulation of peat (i.e. partially decayed dead organic matter) and permanent waterlogging. Bogs are ombrotrophic, meaning they receive their water primarily from precipitation while fens are minerotrophic and have a more variable amount of groundwater input. The two types are further distinguished from each other by the amount of peat accumulation and acidity; bogs have more peat and higher acidity compared to fens.

In contrast to organic wetlands, mineral wetlands (e.g., swamps, marshes, and open water wetlands) are characterized by little to no accumulation of peat (National Wetlands Working Group, 1997). Swamps are characterized by the dominance of trees in their vegetation communities and prolonged periods of saturation (National Wetlands Working Group, 1997). Marshes are nutrient-rich wetlands distinguished from swamps by herbaceous emergent vegetation and periodic flooding. Shallow open water wetlands are characterized by floating or submerged vegetation existing in permanent water bodies.

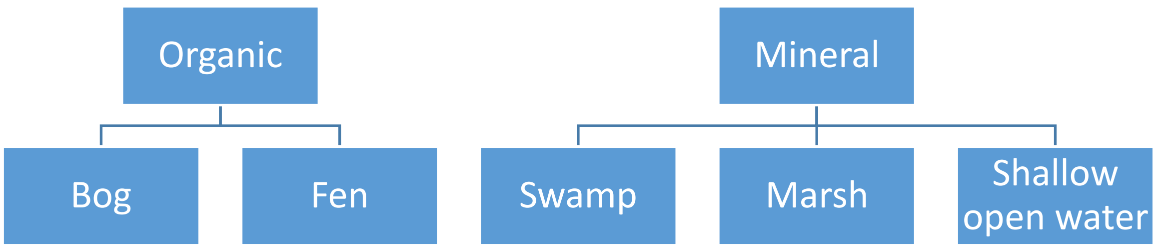

Figure 1.1: Classification of wetlands based on hydrology and nutrient supply

Although it may be useful to have a wetland classification system that is globally applicable, it is important to consider that these five classes do not capture the more nuanced differences between wetland types, which is why more specific classification schemes are also used. For example, in the Prairie Pothole Region (PPR) of North America, wetlands are typically formed in small depressions created by the last glacial retreat with depths usually less than 1 metre, and a median area of 1,600 m^2 (Huang et al., 2011). Most of the PPR wetlands could be classified as marsh or shallow open water, but wetlands in the PPR are more precisely described in terms of their hydroperiod (i.e. ephemeral, temporary, seasonal, semi-permanent, or permanent; Stewart & Kantrud, 1971; Wu & Lane, 2016).

2 – Wetland Function and Services

Wetlands draw keen interest from multiple academic disciplines and this is partially attributable to the fact that wetlands provide numerous ecological and anthropogenic services. One of the most prominent examples of wetland services comes from the role that wetlands play in regional hydrologic cycles where they affect discharge, precipitation, evaporation, transpiration and storage (Wray & Bayley, 2006). Storage is of particular importance because wetland depressions absorb storm water during times of peak flow, which mitigates downstream floods (Ogawa & Male, 1986). Additionally, water storage makes wetlands ideal habitats for a wide of variety of fish species (Uzarski et al., 2005). Wetlands also gradually release water during times of drought, which replenishes downstream water supplies (Keddy, 2010; Woodward & Wui, 2001).

The role of wetlands in biogeochemical cycling is another source of benefits provided to human society. Wetland soils, vegetation and microbial communities have been found to remove excess nutrients and pollutants from surface runoff (Piehler & Smyth, 2011), acting as a natural filtration system in the maintenance of water quality. Engineered wetlands are often constructed to mimic the filtration processes occurring in natural wetlands (Vymazal, 2007). Wetlands (natural and engineered) can remove nitrogen from wastewater primarily through ammonia volatilization (conversion of soil nitrogen into ammonia gas), denitrification (bacterial process that converts nitrate to nitrogen gas), and plant uptake (conversion of inorganic nitrogen to build plant tissue) (Vymazal, 2007). Similar to nitrogen, there are a variety of processes that occur in wetlands which remove phosphorus from water including plant uptake, microbial uptake, soil accretion, and adsorption (Vymazal, 2007).

Wetlands also play an important role in the carbon cycle and subsequently can affect climate change. The role of wetlands in the carbon cycle is complex because they can act as either carbon sinks or sources. They are typically net carbon sinks because the anaerobic soils prevent total decomposition of plant detritus, which leads to a build up of carbon over time (Kayranli et al., 2010).

The amount of carbon storage in wetlands varies depending on wetland class and location. Peatlands provide long-term carbon storage at a ratio well out of proportion with the amount of land they occupy. For example, peatlands in the boreal region of the Canadian prairie provinces (Manitoba, Saskatchewan and Alberta), account for 2.1% of global terrestrial carbon storage while only occupying 0.25% of the land (a storage-landcover ratio of 8.4; Vitt et al., 2000). The capacity for long-term carbon storage is also true at the global scale where peatlands account for 16-33% of the global carbon pool while only covering approximately 3% of the land surface (storage-landcover ratio between 5.3 and 11) (Bridgham et al., 2006). Tropical and temperate wetlands are even more effective at sequestering carbon, typically storing between 4-5 times more carbon on annual basis relative to boreal wetlands (Mitsch et al., 2013).

Despite the long-term carbon storage, the role of wetlands in the global carbon cycle is complicated by the fact that they are also emitters of methane (CH_4) (Belyea & Baird, 2006), which is 25 - 36 times more effective at trapping outgoing long-wave radiation than carbon dioxide (CO_2) (Environmental Protection Agency, 2015; Mitsch et al., 2013). The balance between carbon storage and emission for wetlands varies regionally, and there are also large uncertainties in carbon emission estimates, but it is likely that CH_4 emissions largely offset much of the carbon storage benefit of wetlands on an annual basis (Bridgham et al., 2006). However, wetlands are net carbon sinks in the long term (i.e. 300 years) because the time periods in which carbon is stored in both wetlands and the atmosphere differ from that of CH_4 (Mitsch et al., 2013). The atmospheric residence time of CH_4 ranges between 9-12 years (Mayer et al., 1982) whereas carbon can be stored in wetlands for centuries. However, wetland loss would result in the emissions of centuries worth of stored carbon into the atmosphere.

Wetlands are biologically productive and diverse ecosystems even though they occupy small amounts of land. For example, wetlands occupy 5% of the United States while supporting 31% of the nation's plant species (Silvy, 2012). Similarly, up to 43% of all federally endangered fish and bird species in the United States rely at least indirectly on wetlands for survival (Silvy, 2012). The biodiversity that is supported by wetland habitats is beneficial globally because it supports long term evolutionary adaptation to threats and disturbances (White et al., 1999).

Wetlands also have direct benefits for human health through provisioning ecosystem services such as drinking water, food for agriculture and livestock, and medicinal products (Horwitz & Finlayson, 2011). They also provide cultural ecosystem services such as educational opportunities, inspiration for artistic pursuits, recreational sports (e.g. fishing and hunting), and aesthetic amenities (Horwitz & Finlayson, 2011)

With these services considered, monetary valuations rank wetlands among the most valuable of ecosystems (Moreno-Mateos et al., 2012), even when accounting for the high variability of estimates in financial valuation (de Groot et al., 2012). However, the ability of wetlands to continue providing their services has come under threat from the rapid growth and industrialization of human activity. It has been estimated that 50% of the world's wetlands have been destroyed since the beginning of the 20th century (Russi et al., 2013). In Canada, up to 70% of wetlands have either been destroyed or degraded in settled areas (Ducks Unlimited Canada, 2008). The primary drivers of wetland loss are increases in anthropogenic land use (drainage and conversion; Findlay & Bourdages, 2000; Gibbs, 2000; Houlahan et al., 2006), climate change (Erwin, 2008), and sea-level rise for coastal wetlands (Nicholls et al., 1999). If conservation, restoration, and reclamation efforts are to reverse the global decline of wetland area, policies need to be informed by interdisciplinary knowledge of wetlands to ensure that wetlands can continue providing their services at both global and regional scales.

3 – Landscape Ecology

Wetlands have been examined through a diverse range of disciplines including biology, ecology, chemistry, hydrology, geography, economics and law. However, there is little sense in examining wetlands through only one of these lenses because they are all linked. Similarly, while wetlands can be studied at a variety of scales, it is necessary to include analysis of wetlands at the landscape scale because they do not exist independently of their surroundings. Wetlands, like any ecosystem, are one part of a spatial hierarchy consisting of lower-level entities (e.g. animals, plants, and energy and water flows) and are themselves components of a larger landscape (de Vasconcelos et al., 1993; White et al., 1999).

Wetlands exist on a continuum of connectivity in terms of hydrological and biogeochemical cycles occurring at the landscape level (Leibowitz, 2003). Riparian and coastal wetlands have strong hydrologic links via overland and stream-water flow. The incoming and outgoing flows of water are key determinants in the vegetation types that occur in the wetland, which in turn affect the animal species that interact with ecosystem (Keddy, 2010). Water flow also brings in nutrients and can bring pollutants from upstream human activity to a wetland ecosystem.

Connectivity is less obvious for geographically isolated wetlands (GIW), such as the marshes found in the PPR. GIWs are completely surrounded by upland ecosystems (Tiner, 2003), and thus receive most of their water from spring snow melt occurring in their containing catchments and minor amounts from precipitation. GIWs were excluded from protection of the U.S. Clean Water Act in 2001 on the grounds that their hydrologic isolation meant they played no role minimizing the pollutant loading of federally protected waters, due to the lack of research empirically documenting the hydrologic connectivity of these wetlands (Marton et al., 2015). Subsequently, the notion of GIWs being hydrologically irrelevant was empirically challenged. For instance, GWI complexes on the Texas Gulf Coast are connected to nearby waters through seasonally wet drainage channels, and account for 0-27% of annual watershed runoff (Wilcox et al., 2011). The runoff from the GIWs on the Gulf Coast is intermittent, occurring primarily during pulses caused by catchment overflow during precipitation events (Wilcox et al., 2011). The relatively limited hydrologic connectivity of GIWs enhances their ability to retain nutrients and pollutants, meaning GWIs play an important role in the biogeochemical cycling of watersheds (Cohen et al., 2016; Marton et al., 2015).

Ecological connectivity is also a major reason for studying wetlands at a landscape scale. Dispersal of flora propagules from neighbouring wetland patches is a key factor in the maintenance of the biodiversity for a wetland patch (Galatowitsch & van der Valk, 1996). The lack of dispersal from natural wetlands is one the main factors responsible for the failure of restored and constructed wetlands from fostering comparable levels of species diversity to natural ones (Galatowitsch & van der Valk, 1996). Wetland-obligate bird species are known to move between individual wetlands and are therefore sensitive to the spatial distribution of wetlands (Haig, Mehlman, & Oring, 1998). Landscapes with a higher density of wetlands have a higher abundance of water fowl as well as their predators (Stephens et al., 2005).

In addition to hydrologic, biogeochemical and ecological connectivity, studying wetlands at the landscape scale is also necessary to account for the variation between individual wetland sites. Site specific studies provide highly detailed descriptions of a single site, but to advance our understanding of wetland processes it is necessary to generalize our findings for broader applicability (Hobbs, 1999). The scientific field of landscape ecology can offer valuable insight to wetland research because it is both inherently interdisciplinary and it explicitly links landscape-scale patterns with ecosystem functions and services. A simple working definition of landscape ecology is the “study of the reciprocal effects of spatial pattern on ecological processes” (Risser, 1999). The term ‘landscape ecology’ was coined by German geographer Carl Troll in 1939 as part of his research using aerial photographs to study the interactions between vegetation and the surrounding environment (Troll, 1971; J. Wu, 2006).

The discipline of landscape ecology flourished in the 1970s and 1980s when developments in computers, remote sensing and geographic information system (GIS) technologies made data collection and analysis at landscape scales more practical to undertake (O’Neill et al., 1999; Risser, 1999; Withers & Meentemeyer, 1999). The inclusion of landscape-level analysis in ecological studies has allowed ecologists to frame their research outputs in a manner that is more relevant to policy makers who must make sustainable planning decisions

(Ahern, 1999; Leitão & Ahern, 2002; Li et al., 2010). Wetland management, like the management of other ecosystems, should be managed at the landscape level for it to be effective (Bedford & Preston, 1988).

Defining the precise meaning of 'landscape' is an important consideration when performing an analysis in landscape ecology. A general definition of 'landscape' is an "area in which variables of interest are spatially heterogeneous" (Wu, 2013). However, this definition allows for multiple methods of delineating landscape units (e.g. administrative boundaries, watersheds) and a range of spatial scales. In practice, determining landscape units is typically study-specific (Risser, 1999) although the landscape units should not be arbitrarily chosen. The method of delineating landscape units should be based on the ecosystems of interest. For example, using watersheds as the landscape unit for a wetlands study would be appropriate because hydrologic flows within a watershed influence the distribution of wetlands (O'Neill et al., 1996). However, there are scenarios where administrative boundaries would be more sensible than natural boundaries such as watersheds. For example, administrative boundaries would be more appropriate for a study of urban forests because management decisions regarding these ecosystems will vary across administrative boundaries.

In addition to choosing the most appropriate method for delineating landscapes, care must also be taken when deciding on the spatial extent(s) of analysis. The appropriate spatial extent needs be large enough that landscape patterns can be understood while remaining within the scope of human policy makers. For example, performing a research study at a provincial or state spatial extent is not useful for municipal policy makers because there will be too many variables external to geographic jurisdiction of the policy makers for them to make informed decisions. Studies at very large spatial extents have to make a number of generalizations, which may not be applicable when scaled down to the municipal level. Ultimately, the definition of landscape for a particular study is one of the most important considerations for determining if the research will be able to effectively inform policy.

Once the landscape is defined, researchers can quantify the spatial heterogeneity of a landscape and relate those measurements to ecological, environmental or social processes. A large suite of metrics for quantifying the spatial heterogeneity of landscapes have been utilized in landscape ecology, but there remains the challenge of identifying which metrics best capture the variation in wetland spatial pattern. Previous landscape ecology research has focused on identifying the best representative metrics, though different studies typically identify different metrics (Cushman et al., 2008). Furthermore, very little research has investigated how human disturbance affects wetland spatial pattern. Chapter 2 attempts to fill this gap by answering the questions “what group of metrics can most effectively describe the structure and pattern of wetland landscapes?” and “how do these metrics relate to human disturbance?” The rationale for these questions is to define a spectrum of reference conditions in terms of wetland composition and configuration, similar to that used in bioassessment of pristine landscapes (Bailey et al., 2004; Herlihy et al., 2008); it is expected that landscapes undisturbed by human activity will be quantifiably different than those that are. Chapter 3 situates the findings of Chapter 2 within the broader context of landscape ecology and outlines a framework for wetland reclamation at the landscape scale equivalent to resource extraction operations currently occurring in Alberta, Canada.

Chapter 2 – A Methodology for Relating Wetland Configuration and Composition to Human Disturbance in Alberta

1 – Introduction

Humans have substantially altered natural landscapes for thousands of years (Ruddiman, 2013) with impacts ranging from local to global scales (Foley et al., 2005). These alterations typically entail the conversion of forests, wetlands, and other natural land-cover types to crop, mining, and impervious urban lands. The land-use activities occurring on these lands are undertaken by different actors (e.g. farmers, urban developers, mining companies) often at relatively small spatial scales, but they aggregate to form landscapes that are radically different in appearance and function from undisturbed landscapes (DeFries et al, 2004).

Parallel to the trend of massive land-cover conversion is a growing recognition of the need to reclaim land (i.e. converting developed land back to its original use; Timoney, 2015). Wetlands are one land-cover and ecosystem type that has been identified as economically (Brander et al., 2013; de Groot et al., 2012; Woodward & Wui, 2001), ecologically (Catallo, 1993; Uzarski et al., 2005), and environmentally (Belyea & Malmer, 2004; Chmura et al, 2003; Kayranli et al., 2010; Vitt et al., 2000) valuable, which suggests the need for effective reclamation planning. Since wetlands affect and are affected by their surrounding landscapes (Houlahan et al., 2006; Lopez et al., 2002; Mack, 2006; Mita et al., 2007; Rooney et al, 2012), landscape-level reclamation (i.e. catchment or region) efforts should be informed by knowledge about the linkages between the local properties of individual wetlands and the patterns of the broader landscape to be sustainable (Mairota et al., 2013; Rooney et al, 2015).

Targets for desired ecosystem functionality and landscape structure are required to assess the success of reclamation efforts. Setting reclamation targets is complicated by the fact that wetland functionality and the spatial patterns of surrounding land cover vary along a gradient of human disturbance. While undisturbed landscapes have provided a benchmark or reference condition (Bailey et al., 2004) for comparison, understanding how wetland functions and land-cover patterns vary along a gradient of human disturbance can provide a continuum of targets for reclamation in human-dominated landscapes (Brooks et al., 2004).

Wetland functions and processes are often measured and monitored through multiple response variables (e.g. richness and diversity of aquatic plant species, Albert & Minc, 2004; salinity, Skinner et al, 2001; and macroinvertebrate, amphibian, fish and bird populations, Brazner et al., 2007) while landscape structure and pattern are typically quantified using algorithmic measures of composition and configuration called landscape metrics. Composition metrics are aspatial and include proportional abundance (the proportion of each class relative to the total landscape area), richness (number of different patch types), evenness (abundance of patch types relative to each other), and diversity (composite of richness and evenness) among others (McGarigal, 2014). Configuration metrics quantify the structure and spatial pattern of 'patches' (i.e. areas with relative homogenous biophysical characteristics). Configuration metrics include patch area, edge, shape, core area (the interior of a patch resulting from the creation of an edge buffer); contrast (relative difference among types); and aggregation (spatial clustering of patches) among others (Cushman et al, 2008).

This paper will use landscape metrics to quantify wetland composition and configuration to answer the question how does wetland composition and configuration vary with changes in human disturbance in landscapes? To answer this question, we suggest a parsimonious set of representative metrics that facilitate the inclusion of landscape pattern in the reclamation design process.

1.1 – Overview of Landscape Metrics and Human Disturbance

Landscape metrics have been used to quantify the characteristics of landscapes along a gradient of wild to human-dominated landscapes. A number of metrics have previously been compared to human disturbance (e.g., urban, agricultural) in landscapes. For example, mean forest-wetland patch-area was negatively correlated with disturbance for watersheds in Pennsylvania (Miller et al, 1997). This is expected because overall forest area decreases with the increasing human land use, and the remaining forest patches will be smaller more fragmented. Conversely, Shannon's diversity index had a positive relationship with human disturbance, explained by the fact the natural watersheds were dominated by forest while the developed watersheds had a mix of forest, residential, and agricultural land (Miller et al., 1997).

Landscape metrics have also been used in temporal analyses with human disturbance due to the availability of time-series remotely sensed data. Temporal analyses provide additional insight relative to single time-step analyses by quantifying how landscape composition and configuration change with increasing human disturbance over time. For example, while quantifying the impacts of urban expansion on wetlands in Jiangsu, China, from 2000 to 2006, Li et al. (2010) observed an increase in wetland landscape fragmentation that was correlated with urban expansion, which is typical of other studies quantifying the impacts of urban expansion and human disturbance (Griffith et al., 2003; Luck & Wu, 2002).

1.2 – Consideration of Spatial Scale

Spatial scale (i.e. spatial extent and data resolution) are known to influence the values of collected ecological data and the subsequent insights about processes drawn from the data analysis (Wiens, 1989). Similarly, spatial scale also influences the behaviour and interpretation of landscape metrics (Wu, 2004). For example, coarse data resolutions tend to positively skew patch size distributions for highly fragmented landscapes while relatively homogenous landscapes (e.g. forested areas) can be accurately represented (O'Neill et al., 1996; Turner et al., 1989). Data resolution also affects metrics describing shape complexity (Cain et al., 1997) because the perimeter and area of a feature may increase and be simplified as resolution becomes more coarse. Spatial extent has a positive relationship with dominance and contagion measures though these metrics are constrained by the number of land-cover classes and the proportion of the landscape occupied by each land-cover class (Turner et al., 1989). The choice of spatial extent must be at least partly dictated by data resolution because spatial heterogeneity can only be quantified when the extent is significantly larger than the minimum mapping unit (Plexida et al, 2014). The appropriate spatial extent for a landscape-pattern analysis should capture the heterogeneity of the landscape with a given data resolution and the extent at which the ecological processes of interest occur (Gustafson, 1998).

The impact of spatial scale on the interpretation of landscape metrics is illustrated by comparing the differences in the relationships between patch shape complexity (i.e. fractal dimension) and urban area observed by Li et al. (2010) and O'Neill et al. (1988). Li et al. (2010) observed an increase in overall shape complexity with urbanization while O'Neill et al. (1988) observed a negative association with urban area. This difference can possibly be explained by the fact that human disturbance was examined for vastly different spatial extents (city vs. region) and spatial resolutions (30 m vs 200 m). Li et al. (2010) conducted a city-scale study with a classification based on higher resolution remotely sensed imagery, which enabled a more detailed representation of the city area. Conversely, O'Neill et al. (1988) conducted a regional study (i.e. large extent and low spatial resolution) that represented urban areas in a highly simplified form, only capturing the extents of urban areas while missing the diversity and complexity contained within.

The thematic resolution (i.e., number of land-cover classes) also affects landscape analysis outcomes (Bailey et al., 2007). A more detailed classification scheme has less thematic aggregation, so landscapes are likely to appear more heterogeneous, and patches will be smaller and more distributed compared to a classification system with lower thematic resolution. For example, Bailey et al. (2007) explicitly tested the effects of thematic resolution on metric selection for European agricultural landscapes and found that dominance (i.e. largest patch index) distinguished landscapes mapped with coarse thematic resolutions, while shape, aggregation and diversity metrics provided more information for finer thematic resolutions.

2 – Methods

2.1 – Study Area

The presented research is situated in the Grassland, Parkland and Boreal Natural Regions of Alberta, Canada (Downing & Pettapiece, 2006) and is constrained by the spatial extent of the Central and Southern wetland inventories (Figure 2.1). Natural Regions are the largest spatial units in Alberta's ecological land classification system. Classes are delineated primarily by climate, soil, vegetation and physiography (Table 2.1; Downing & Pettapiece, 2006).

The three Natural Regions (i.e. Grassland, Parkland and Boreal) within which our study area is situated occupy 81% of Alberta (Downing & Pettapiece, 2006), which means they encompass much of the province's climatic, ecological and biophysical variability. The primary land uses in the Grassland and Parkland regions are till cropping, grazing, recreation, and oil and gas extraction (Downing & Pettapiece, 2006). The southern parts of the Boreal region also have some till cropping activities but recreation and resource extraction (i.e. forestry, mining, oil and gas) are the dominant land uses further north (Downing & Pettapiece, 2006).

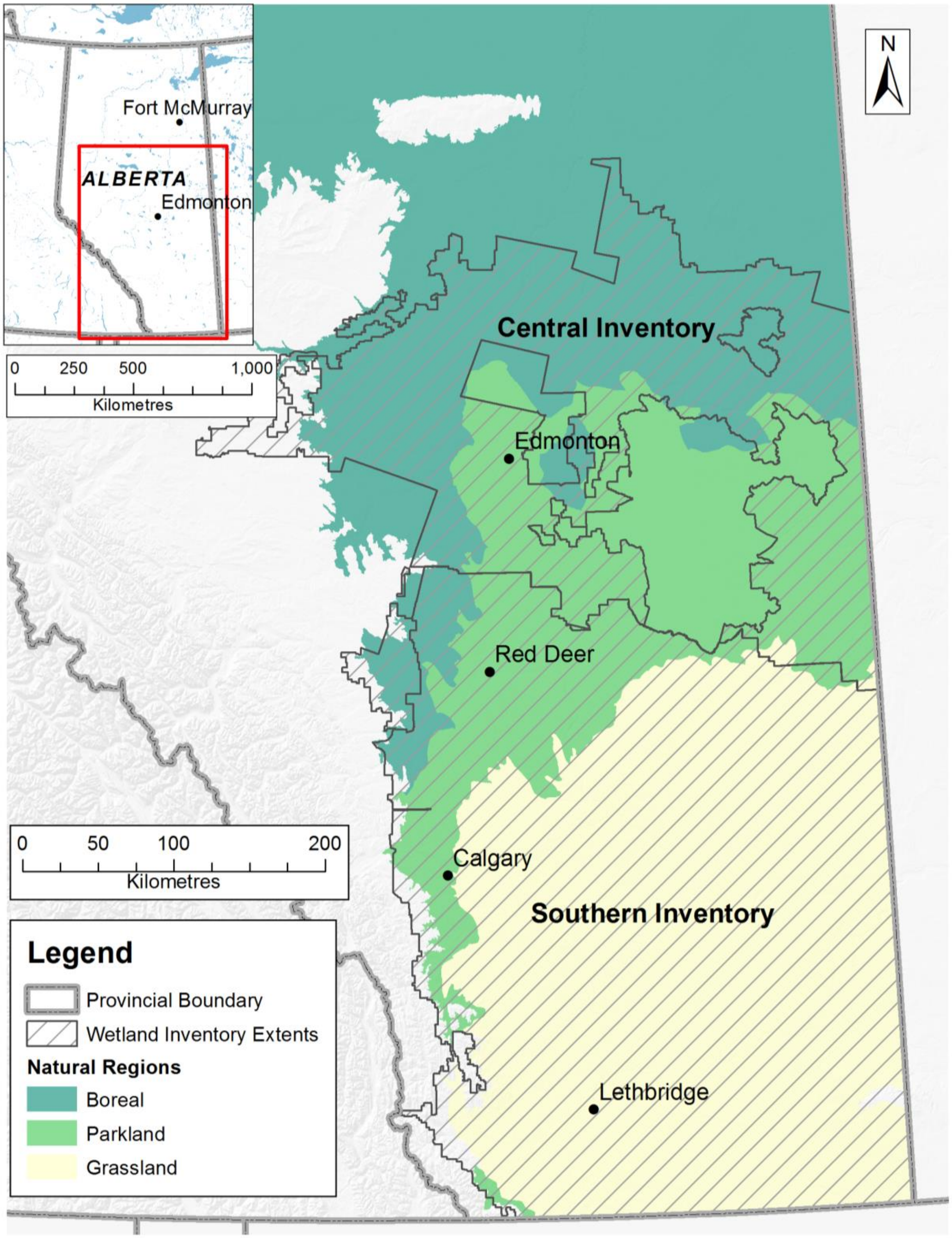

Figure 2.1: Map of the study area as defined by the extents of the Central and Southern wetland inventories overlapping with the Boreal, Parkland and Grassland Natural Regions of Alberta, Canada.

Table 2.1: Climate characteristics for the Natural Regions of interest (1961-1990). Adapted from *Natural Regions Committee (2006)*. Average summer moisture index (SMI) is the area-weighted average of the SMI values for the Subregions within each Natural Region.

Natural Region	Area (km^2)	% of province	Mean annual temp (°C)	Mean annual precipitation (mm)	Growing degree days >5°C (GDD5)	Average summer moisture index (SMI)	Soil	Dominant vegetation	% wetland area
Grassland	95,565	14	4.0	374	1,592	6.03	Brown chernozems, brown solonetz, and gleysols.	Grass, shrub, and fescue	4
Parkland	60,747	9	2.3	447	1,391	4.23	Dark grey to black chernozems, solonetzic, and gleysols	Grass, fescue, aspen. Tree cover increases with latitude	9
Boreal	381,046	56	-0.2	469	1,207	3.84	Orthic gray luvisols, bruvisols. Wetlands are mesisols, fibrisols and gleysols	Aspen, mixewood, jack pine, black spruce, peatland	35

The presented research is primarily situated in southern Alberta (i.e. Grassland and Parkland) despite the large-scale extraction activities in the Boreal region (e.g. Athabasca and Peace River Oil Sands). The study area was chosen with purpose to guide the long term (i.e. decade-century) reclamation period required to obtain legal closure permits for these activities. Over this time period it is expected that climate change will cause the Grassland and Parkland regions to shift northward into what is currently Boreal (Schneider, 2013). Therefore, to guide future policy and industrial reclamation efforts, it is necessary to analyze the Grassland and Parkland regions to prepare for future conditions (Rooney et al., 2015).

Within the study area, analysis was performed at 'reclamation scale', which in this context is analogous to the disturbance footprints associated with individual *in situ* oil extraction operations in Alberta. Disturbance footprints encompass gravel pits, bitumen treatment plants, steam generators, well pads, worker living quarters, and water treatment plants. Based on measurements made from satellite imagery of oil extraction operations, reclamation scale was estimated to have an average spatial extent of 1 km^2. However, this study's methodology can be applied to reclamation projects of different spatial extents.

2.2 – Data

The presented research was constrained by the spatial coverage of two non-overlapping wetland inventories (Figure 2.1) covering Central and Southern Alberta. The Central inventory covers the northern and western parts of the Parkland and southern section of the Boreal (Figure 2.1). The section of the Boreal that is covered by the Central inventory consists of the Central Mixedwood and Dry Mixedwood Subregions. These Subregions are warmer than the Boreal Forest average (Table 2.1) with mean annual temperatures of 0.2°C and 1.1°C, respectively (Downing & Pettapiece, 2006). However they have mean annual precipitation levels that are comparable to the Boreal average (Downing & Pettapiece, 2006). The Southern inventory covers all of the Grassland and the southwestern part of the Parkland (Figure 2.1). It should be noted that the central portion of the Parkland is not covered by either inventory. Despite this large spatial gap, 34% of the Parkland is covered by the Central inventory and 33% is covered by the Southern inventory.

The Central and Southern wetland inventories both use the Alberta Grassland Vegetation Inventory (GVI) classification system which delineates lentic wetlands based on hydroperiod (Alberta Sustainable Resource Development, 2011). The classification system consists of 5 distinct classes: temporary, seasonal, semi-permanent to permanent, open water, and alkali (Table 2.2). These classes are analogous to classes II – VI from the Stewart & Kantrud (1971) classification system. Lotic wetlands (i.e. riverine wetlands) are not considered within these classes.

Table 2.2: Descriptions of the lentic wetland permanence classes used for the Central and Southern wetland inventories (Alberta Sustainable Resource Development, 2011).

Class	Description
Temporary	Surface water retained only briefly after spring melting period. Low-prairie and wet-meadow vegetation.
Seasonal	Surface water retained for more than three weeks. Lusher vegetation relative to Temporary wetlands due to higher water table.
Semi-permanent to permanent	Surface water persists except in times of extreme drought. Emergent vegetation (e.g. cattails, bulrushes).
Open water	Permanent open-water areas larger than 1 ha.
Alkali	Surface water retained between a few weeks to a few months. Minimal vegetation. Saline crust.

Both wetland inventories were provided by Alberta Sustainable Resource Development (ASRD); however, different methods were used in the creation of the inventories. The Central inventory was constructed using SPOT 5 imagery (2006-2009), a 25 m resolution digital elevation model (DEM) and ancillary data such as roads and hydrography line features (Alberta Sustainable Resource Development, 2010). The imagery was segmented into homogenous units with the size of each unit ranging from 0.001 ha to 2.0 ha. An object-based classification was then performed on the image segments to delineate land covers. Then a predictive ecosystem decision-tree model was used to produce the final wetland classifications (Alberta Sustainable Resource Development, 2010).

The accuracy of the Central inventory was assessed using 100 randomly selected 2 km x 2 km assessment zones. SPOT 5 and orthorectified aerial imagery (orthoimagery) were manually interpreted for each zone and compared to the automated classification. The observed and expected accuracies of the classification were 83% and 64% respectively ($\kappa = 0.51$) (Alberta Sustainable Resource Development, 2010).

The Southern inventory was created using SPOT 5 imagery (2006-2008), orthoimagery (2005-2006), and SPOT 4 imagery (2006-2008) where cloud cover prohibited the classification of SPOT 5 imagery (Alberta Terrestrial Imaging Center, 2009). A supervised classification of temporally stacked imagery was performed using a support vector machine algorithm to identify wetland boundaries. The minimum mapping unit was 0.2 ha (20 SPOT pixels) and wetlands less than 0.2 ha in area were excluded. A second classification was then performed to classify the wetlands by GVI classes (Table 2.2). The wetland boundary polygons were classified by overlaying the wetland class image and selecting the dominant GVI class within the boundary.

Accuracy assessment for the Southern inventory was performed within 5 township boundaries where ground-truth data were produced by manually digitizing and classifying wetlands from aerial imagery with cell sizes between 0.5 m and 2.5 m. The percent of correctly identified wetland boundaries ranged from 67% to 85% between the townships while the accuracy of the subclasses ranged from 51% to 68%. These accuracy measurements illustrate that the Southern inventory has a lower wetland classification accuracy than the Central inventory.

In addition to the wetland classification data and natural region boundaries provided by ASRD, Agriculture and Agri-Food (AAFC) annual crop inventory data for 2009 were acquired. The annual crop inventory maps crop types and other land covers (e.g., wetland, forest and developed areas) across Canada's arable region at a 56 m resolution. Annual crop inventory data are produced using a decision tree method applied to a combination of Landsat-8 and RADARSAT-2 imagery. Results are ground-truthed using data provided by crop-insurance companies and they have a minimum accuracy of just under 90% for agricultural land (accuracies for other land covers are not reported). The crop inventory from 2009 was used because this was the closest year to the SPOT image dates (2006-2009) used to create the wetland inventory. The crop inventory dataset included 22 agricultural classes and 9 non-agricultural classes (3 of which were for forest) for the study area. It was assumed that the agricultural and forest land covers had similar effects on wetlands so the crop inventory was reclassified into the following more general land-cover classes: water, exposed, developed, shrub, wetland, grass, agriculture, and forest (Table 2.3).

Along with the crop inventory data, a 1:20,000 DEM covering Alberta at a 10 m resolution was acquired from Alberta Innovates Technology Futures (AITF) as well as watershed boundary data from Alberta Environment and Parks. Watershed boundaries are defined by the Water Survey of Canada (WSC) and were delineated in 1998-1999 using 1:50,000 topographic maps with some ancillary aerial photography of varying scales (Alberta Environment and Sustainable Resource Development, 2014). These received minor updates in 2010-2011 and 2014. The WSC regions approximate level 6 of the nested hierarchical Hydrologic Unit Code (HUC) classification system originally developed by the United States Geological Survey (Alberta Environment and Sustainable Resource Development, 2015; Seaber et al, 1987).

Table 2.3: Description of the aggregated landcover classes in the AAFC crop/landcover inventory (Agriculture & Agri-Food Canada, 2014)

Landcover	Description
Water	Water bodies (lakes, reservoirs, rivers, streams, salt water, etc.).
Exposed	Land that is predominately non-vegetated and non-developed. Includes glacier, rock, sediments, burned areas, rubble, mines, and other naturally occurring non-vegetated surfaces. Excludes fallow agriculture.
Developed	Land that is predominantly built-up and vegetation associated with these land covers. This includes road surfaces, railway surfaces, buildings and paved surfaces, urban areas, industrial sites, mine structures, etc.
Shrub	Predominantly woody vegetation of relatively low height (generally < 2 m). May include grass or wetlands with woody vegetation, regenerating forest.
Wetland	Land with a water table near/at/above soil surface for enough time to promote wetland or aquatic processes (semi-permanent or permanent wetland vegetation, including fens, bogs, swamps, sloughs, marshes).
Grassland	Predominantly native grasses and other herbaceous vegetation, may include some shrubland cover.
Agriculture	Agricultural land, including annual crops, perennial crops and pasture; excludes native grassland
Forest	Predominantly forested or treed areas (> 2 m). Includes both deciduous and coniferous.

2.3 – Analysis

The analysis to identify representative metrics and relate them to human disturbance was split into 5 conceptual steps (Figure 2.2). Broadly speaking, the method involves 1) defining the landscapes to be analyzed, 2) preparing the wetland and AAFC land cover data, 3) calculating the landscape metrics for each analysis landscape, 4) reducing the original set of metrics through variable reduction procedures, and 5) comparing metric values to human disturbance at the 1 km^2 reclamation scale.

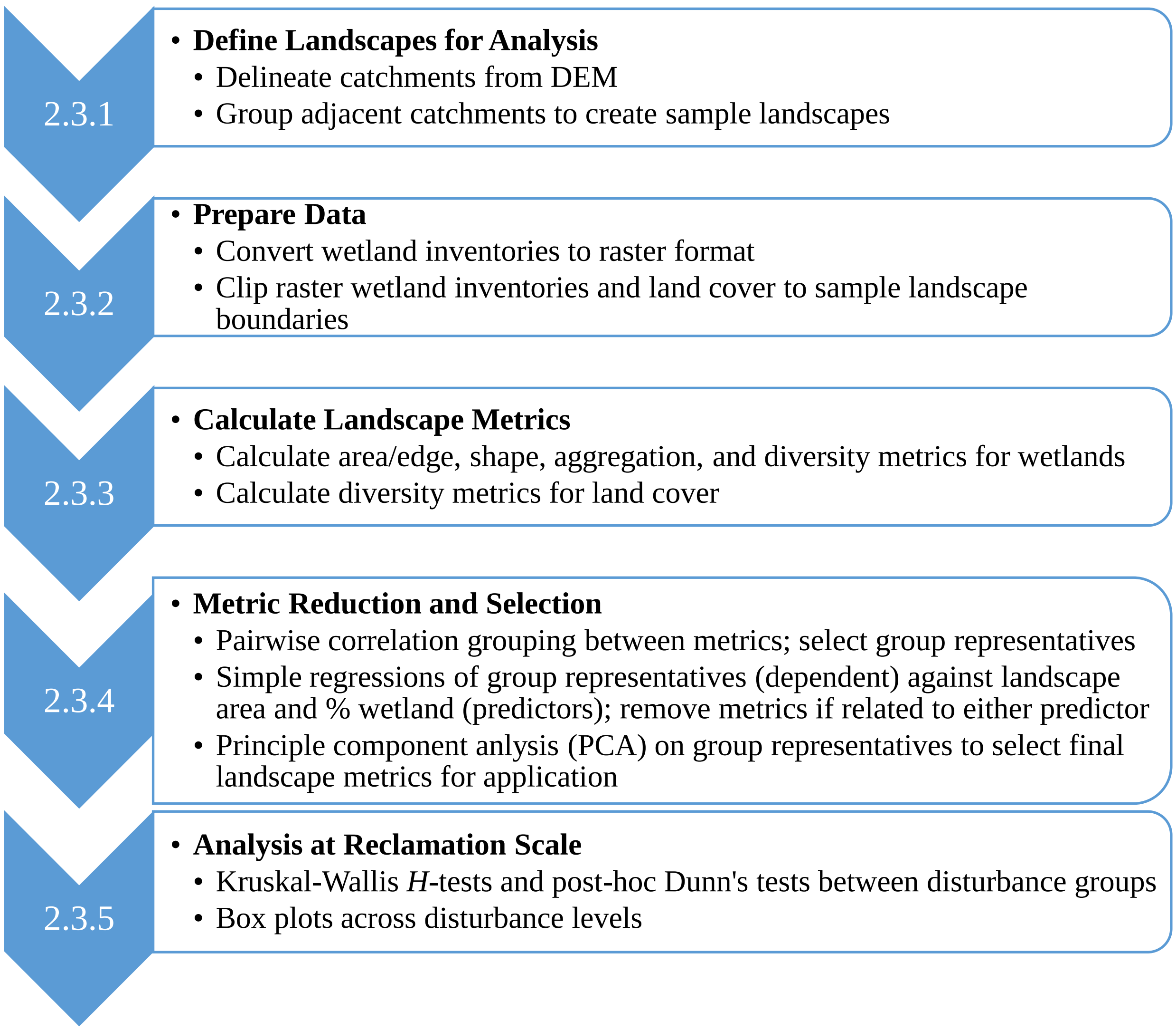

Figure 2.2: Overview of analysis divided into 5 steps. Each step corresponds to text subsection numbering

2.3.1 – Define Landscapes for Analysis. The first step of the analysis was to define hydrologically meaningful landscapes for assessment. Hydrologically meaningful units are defined based on the flow direction and accumulation of surface water (i.e. drainage catchments where all surface water flows converge to a single point). Wetland catchments were created in ArcGIS using the DEM (10 m cell size). The average area of the generated catchments was 0.05 km^2, which was 20 times smaller than the extent of the defined reclamation scale of 1 km^2. To accommodate for large reclamation projects (including roadways, well pads, central processing facilities, and mining pits), selected catchments were merged with adjacent catchments and their adjacent neighbours (i.e., spatial lag of 2) to create sample landscapes for analysis.

Sample landscapes in the Central inventory were chosen subject to four constraints: wetland inventory accuracy, presence of cloud cover in the inventories' source imagery (SPOT), proximity to SPOT scene boundaries, and proximity to WSC boundaries. Only sample landscapes located within SPOT scenes with an average accuracy greater than 80% were retained. If these sample landscapes overlapped with areas identified as cloud cover, they were excluded due to uncertainty in permanence classification (Alberta Sustainable Resource Development, 2010). If sample landscapes crossed SPOT scene boundaries they were excluded, since each scene was independently classified and differences in accuracy exist among scenes. Similarly, sample landscapes that crossed WSC borders were excluded to ensure each landscape was within the same hydrological system. In contrast to the Central inventory, a detailed accuracy assessment and data regarding the presence of cloud cover and SPOT scene locations for the Southern inventory were not available. In the absence of these data, sample landscapes in the Southern inventory were only excluded if they intersected WSC borders.

In total, 1,000 random sample landscapes were retained in each of the Central and Southern wetland inventories. The average area of the sample landscapes was 2.0 km^2 (n=1,000; standard deviation: 3.4 km^2) and 2.1 km^2 (n=1,000; standard deviation: 1.9 km^2) for the Central and Southern inventories, respectively. The frequency distributions of the sample landscapes for both inventories were right skewed with mean area within the 95^{th} percentile at 1.6 km^2 and 1.8 km^2 for the Central and Southern inventories, respectively. The sample landscapes approximated the size of a large disturbance and surrounding area as well as maintained the hydrological integrity of the landscape that affects wetland function.

2.3.2 – Prepare Data. The second step involved pre-processing wetland inventory and the AAFC data and extracting those data for each sample landscape. Since many landscape metrics require at least one wetland patch to be present, sample landscapes with no wetland patches were removed. The final sample size was 942 sample landscapes for the Central inventory and 840 in the Southern inventory (1,782 total).

The wetland inventories were constructed by ASRD as vector polygon data but were converted to raster data for calculating landscape metrics (Fragstats; McGarigal, Cushman, & Ene, 2012). The raster wetland inventory data had a 10 m resolution to coincide with the multispectral SPOT imagery from which they were created. All wetland and land cover raster data were given one-cell borders around the sample landscape boundaries with a numeric code used to distinguish between true patch edges and sample-area edges (McGarigal, Cushman, & Ene, 2012).

2.3.3 – Calculate Landscape Metrics. The third step involved calculating the landscape metrics for the sample landscapes. The calculated metrics can be conceptually divided into 4 types: area/edge, shape, aggregation, and diversity/evenness (McGarigal et al., 2012). Area/edge and shape metrics are initially calculated for individual patches (e.g. radius of gyration) and then summarized over a landscape as an area-weighted mean (AM). Unlike area/edge and shape, the aggregation and diversity/evenness metrics are only calculated for all wetland types or land-cover types in the sample landscape (e.g. proportion of like adjacencies).

In addition to calculating/summarizing metrics over the entire sample landscape, three metrics (edge density, patch area, and patch density) were calculated/summarized by GVI wetland class. These three metrics were applied to wetland classes individually since many sample landscapes are missing at least one wetland class and these metrics may still be calculated and render meaningful results in the absence of a given wetland class, whereas other metrics would not (e.g., fractal dimension).

The AAFC crop inventory data provides a landscape matrix that situates wetlands as one among eight different land-cover types. These data enabled the calculation of diversity/evenness metrics that describe the range of different land-cover types and their proportional contribution to each sample landscape. While it would have been possible to calculate other metrics using these data, the resolution of the AAFC data was more granular (56 m) which not only reduced the boundary accuracy of the mapped wetlands but also affected their positional accuracy, which was not measured by AAFC.

In total, 47 landscape metrics were calculated; 40 of these were from the wetland inventory data and 7 diversity/evenness metrics were derived from the AAFC crop inventory data. In addition to these 47 metrics, calculations were made for percent area of each wetland class (i.e. temporary, seasonal, semi-permanent, open water, and alkali), and for wetlands as an aggregated class.

2.3.4 – Metric Reduction and Selection. The fourth step was to reduce the 47 calculated landscape metrics to a set of non-correlated metrics independent of landscape area and composition. To check the robustness of the metric selection across different datasets and Natural Regions, a variable reduction approach was run separately for the two inventories (Central-All and Southern-All) and the Natural Region subsets within the inventory extents (Central-Boreal, Central-Parkland, Southern-Parkland and Southern-Grassland).

The first part of the variable reduction process was to group the landscape metrics by pair-wise Pearson correlation coefficients to reduce collinearity (Moreno-Mateos, Mander, Comín, Pedrocchi, & Uuemaa, 2008; Plexida et al., 2014; Riitters et al., 1995). A matrix of all pair-wise Pearson correlation coefficients between metrics was generated and then metrics were grouped together if the absolute values of all pairwise coefficients were greater than 0.9, as done by Riiters et al. (1995). A single metric was then chosen for each group based on interpretability. To ensure that the selected group representatives were not highly correlated, a second iteration of the correlation procedure was run with only the group representatives using the same Pearson coefficient threshold of 0.9.

After the correlation grouping, the number of metrics was further reduced through a series of simple regression analyses (i.e. one predictor variable). Since it has been observed that metrics can predictably vary with total landscape area (Herzog, Lausch, Thulke, Steinhardt, & Lehmann, 2001) and/or land-cover proportion (Cushman et al., 2008; Long, Nelson, & Wulder, 2010; Mairota et al., 2013; McGarigal & McComb, 1995; Remmel & Csillag, 2003), metrics were regressed against these two predictor variables separately. For the land-cover proportion regressions, wetland class metrics were regressed against the percent area of the corresponding wetland class while metrics applied to the entire sample landscape were regressed against total percent area in wetlands. Linear, quadratic, cubic, and exponential regression models were applied. Metrics were removed if the r^2 value of at least one of the models was greater than 0.2. This threshold is subjective but was strict enough to remove influences of landscape area and composition on metric values.

Lastly, a principle component analysis (PCA; R Core Team, 2015), using a correlation matrix, was run for the metrics retained after the regression analyses to establish a new set of orthogonal variables, called "components", ordered (descending) by their explained variance (Herzog et al., 2001; Mladenoff, Niemi, & White, 1997; Moreno-Mateos et al., 2008; Riitters et al., 1995). Similar to what has been done in bioclimate envelope modelling (Metzger et al., 2013), the most important metrics were identified by the strength of their factor loadings. A representative metric was determined for each principal component with an eigenvalue greater than 1 by selecting the metric with the highest absolute loading on the component. Typically, only one metric was selected for each component although if metrics from different categories (area/edge, shape, aggregation, diversity/evenness) had comparably high loadings, both were selected.

2.3.5 – Analysis at Reclamation Scale. The fifth step was to relate the identified metrics to varying levels of human disturbance at reclamation scale. A new set of 2,000 random points was generated within each of the Central and Southern inventory extents (4,000 total). These points were used as the centroids for generating 4,000 new 1 km^2 square sample landscapes. Square sample landscapes not containing a single wetland patch were filtered out before metric calculation, leaving 1,912 in the Central inventory and 1,522 in the Southern inventory (3,343 total). The metrics identified for the Central and Southern landscapes in Step 4 (Section 2.3.4) were then calculated for the new square sample landscapes.

Anthropogenic disturbance was quantified as the percent of the square sample landscapes occupied by either developed or agriculture land covers. Square sample landscapes were classified into five equal twenty-percent disturbance intervals and the distributions of metric values for each group were qualitatively compared with boxplots. Quantitative comparisons of the metric distributions among disturbance intervals were done with the Kruskal-Wallis *H* test, a non-parametric measure of stochastic dominance among groups (Kindscher, Fraser, Jakubauskas, & Debinski, 1998; Kruskal & Wallis, 1952). The Kruskal Wallis *H* test is similar to an analysis of variance (ANOVA) except that it does not assume data normality, though it still requires that variances be similar among groups (Elliott & Hynan, 2011). Assuming there are no tied observations, H is calculated as

$$H = \frac{12}{N(N+1)}\sum_{i=1}^{C}\frac{R_i^2}{n_i} - 3(N+1) \tag{1}$$

where N is the number of observations in all samples combined, R_i is the sum of the ranks in the sample i, n_i is the number of observations in sample i, and C is the number of samples. In cases where ties occur, each observation is given the mean rank of the tie and Equation (1) is modified to

$$H = \frac{\frac{12}{N(N+1)}\sum_{i=1}^{C}\frac{R_i^2}{n_i} - 3(N+1)}{1 - \frac{\sum T}{N^3 - N}} \tag{2}$$

where the summation of T is calculated over all groups and each T is calculated as

$$T = t^3 - t \tag{3}$$

and t is the number of tied observations in the group.

H follows a chi-squared distribution where larger values indicate the difference between at least two of the groups assessed is statistically significant (Kruskal & Wallis, 1952). The associated p-values along the chi-square distribution indicate the probability that the mean ranks of the groups are the same.

The Kruskal-Wallis test is a useful omnibus test but it does not identify which specific group is significantly different from each other group. When pairwise comparisons of metrics distributions in different disturbance groups was required, the post-hoc Dunn's test (Dunn, 1964) was used. To perform the Dunn's test, a z-score is calculated between the mean ranks of the two groups being compared (retaining the same ranks as the Kruskal-Wallis omnibus analysis; Dinno, 2015). The z-score for comparing groups A and B is calculated as

$$z_{AB} = \frac{y_{AB}}{\sigma_{AB}} \tag{4}$$

where y_{AB} is the difference in mean ranks for groups A and B, and σ_{AB} is the standard error of y_{AB} which is calculated as

$$\sigma_{AB} = \sqrt{\left[\frac{N(N+1)}{12} - \frac{\sum T}{12(N-1)}\right]\left[\frac{1}{n_A} + \frac{1}{n_B}\right]} \tag{5}$$

where n is the number of observations in sample group. The p-values are determined from the area under the normal distribution curve for the calculated z-score. To minimize the likelihood of identifying Type I errors when making a large number of comparisons, a Bonferroni correction was applied where the Dunn's test p-values were multiplied by the number of comparisons made.

To summarize, metric distributions were compared using boxplots, Kruskal-Wallis tests, and pair-wise Dunn's tests, across disturbance intervals for both Central and Southern landscapes as well as between Natural Regions within each wetland inventory.

3 – Results

3.1 – Variable Reduction

Representative metrics were identified by performing a series of variable reduction techniques on six spatial subsets (i.e. Central-All, Central-Parkland, Central-Boreal, Southern-All, Southern-Parkland, and Southern-Grassland). The first step, correlation grouping, removed between 14 to 17 of the original 47 metrics for each spatial subset (Appendix A). The second iteration of the grouping procedure, run only on the representatives from the first iteration, identified strong linear correlations between Shannon's diversity (SHDI), and Simpson's diversity (SIDI) and evenness (SIEI) representatives, while all other pairwise Pearson coefficients were below the threshold of |0.9|. Ultimately, SIDI was selected as the representative diversity/evenness index in each subset because of the simplicity in interpreting its values as proportions.

The next step of the variable reduction procedure removed metrics related to either total landscape area or percent wetland in the landscape. The regression models relating the metrics to landscape area resulted in 2 to 7 metrics being removed, and those relating metrics to percent wetland resulted in 16 to 23 metrics being removed. All of the area/edge metrics were removed from all subsets based of their associations with percent wetland (unless they had already been removed due to associations with landscape area). Additionally, all of the patch density (PD) metrics were removed except PD of open water (PD_OW) in the Central-Boreal subset. Of the remaining shape metrics (i.e., CIRCLE and SHAPE), SHAPE was removed due to a moderate association with percent wetland, while CIRCLE was retained for all subsets. All diversity/evenness metrics (for both land cover and wetland types) were retained in all subsets. In summary, between 7 to 8 of the original 47 metrics remained for each spatial subset following the regression analyses and these were predominantly aggregation and diversity/evenness metrics.

The PCA analyzed between 7 to 8 axes (i.e. the number of metrics) for each of the six spatial subsets. The metric with the highest factor loading on each component with an eigenvalue greater than 1 was selected (Table 2.4; see Table 2.5 for full metric names) for all subsets except for Central Boreal, where two metrics (SIDI_LAND and PD_OW) were selected for the third component due to nearly equivalent factor loadings (Table 2.4; see Appendix B for full factor loadings tables).

Table 2.4: Final representative metrics identified through principle component analysis (PCA). A PCA was run for each subset and the representatives were the ones with the highest factor loading(s) on each of the components with an eigenvalue greater than 1.

		Component			
Inventory	**Subset**	1	2	3	4
Central	**Central All**	SHAPE_WET shape	SIDI_WET diversity/ evenness	SIDI_LAND diversity/ evenness	N/A
	Central Boreal	SHAPE_WET shape	SIDI_WET diversity/ evenness	SIDI_LAND diversity/ evenness PD_OW aggregation	ENN_WET aggregation
	Central Parkland	COHES_WET aggregation	CIRCLE_WET shape	SIDI_WET diversity/ evenness	N/A
Southern	**Southern All**	COHES_WET aggregation	CONTAG_WET aggregation	CIRCLE_WET shape	SPLIT_WET aggregation
	Southern Parkland	COHES_WET aggregation	AI_WET aggregation	SIDI_LAND diversity/ evenness	N/A
	Southern Grassland	COHES_WET aggregation	CONTAG_WET aggregation	CIRCLE_WET shape	SPLIT_WET aggregation

The PCAs produced 3 to 4 representative metrics for each spatial subset (22 total; Table 2.4). Aggregation (11) metrics were the most common due to the higher frequency of that type retained after previous reduction steps. Of the 22 metrics, 10 were unique (Table 2.5). Of the 10 unique metrics, only 3 occurred at least once in both inventories (COHES_WET, SIDI_LAND, CIRCLE_WET). Furthermore, different aspects of wetland and landscape pattern were emphasized between inventories. The Central inventory had a relatively even distribution of shape, aggregation and diversity/evenness metrics while the Southern inventory metrics are mostly aggregation. Furthermore, the Parkland subsets of the Central and Southern inventories only had COHES_WET in common.

Table 2.5: Description of all final representative metrics. See Appendix C for metric formulas, units, and ranges.

Type	Metric	Abbreviation
Shape	Area-weighted mean related circumscribing circle of wetland patches	CIRCLE_WET
	Area-weighted mean shape index of wetland patches	SHAPE_WET
Aggregation	Aggregation index for wetlands	AI_WET
	Patch cohesion index for wetlands	COHES_WET
	Contagion index for wetlands	CONTAG_WET
	Euclidean nearest neighbour of wetland patches	ENN_WET
	Patch density of open water	PD_OW
	Splitting index for wetlands	SPLIT_WET
Diversity/ Evenness	Simpson's diversity index of wetland classes	SIDI_WET
	Simpson’s diversity index of land cover	SIDI_LAND

In addition to the contrast between selected metrics for the two inventories, there were differences between Natural Region subsets within the same inventory. The Central Boreal and

Central Parkland subsets only have SIDI_WET in common. Both of these subsets have metrics quantifying wetland shape and aggregation though the specific metrics differ. Similarly, the Southern Parkland and Southern Grassland subsets only have COHES_WET in common; the other measures of aggregation differ. Furthermore, even when relaxing the stringency of selecting the representatives by taking any metric with an absolute loading greater than 0.5 on each component, there were still few common representatives between Natural Regions. Under the relaxed selection criteria, only 4 of 9 unique metrics were identified as representatives in both Central Boreal and Central Parkland, and 2 of 7 unique metrics were identified in both Southern Parkland and Southern Grassland (Appendix D).

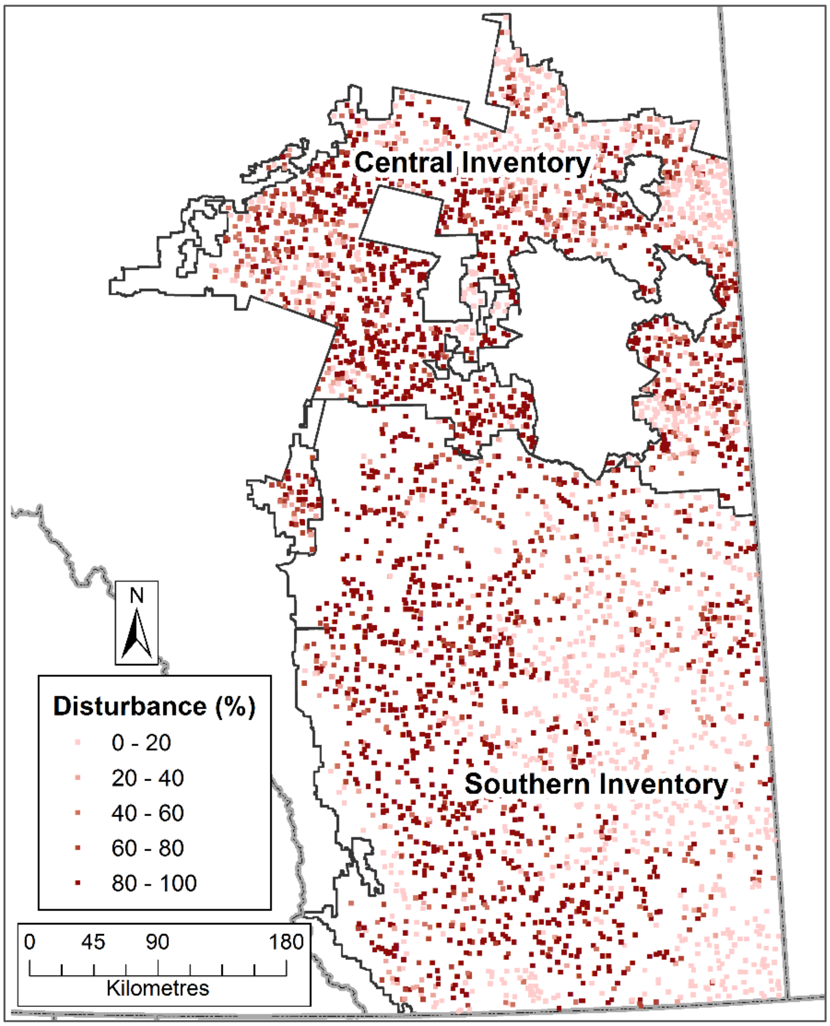

Figure 2.3: Map of disturbance intervals for the 1 km^2 square landscapes. Disturbance was calculated as the total percentage of the landscape occupied by either urban or agricultural land. The landscape squares have been enlarged to enhance visibility.

3.2 – Relating Metrics to Human Disturbance

Disturbance levels of the 1 km^2 square sample landscapes were predominantly in either the lowest (0 – 20%) or highest (80 – 100%) disturbance intervals (Figure 2.3). For the Central inventory 67% of the 1,912 total observations fall within either the lowest or highest disturbance intervals. Similarly, 75% of the 1,522 landscapes in the Southern inventory were in either the lowest or highest disturbance intervals.

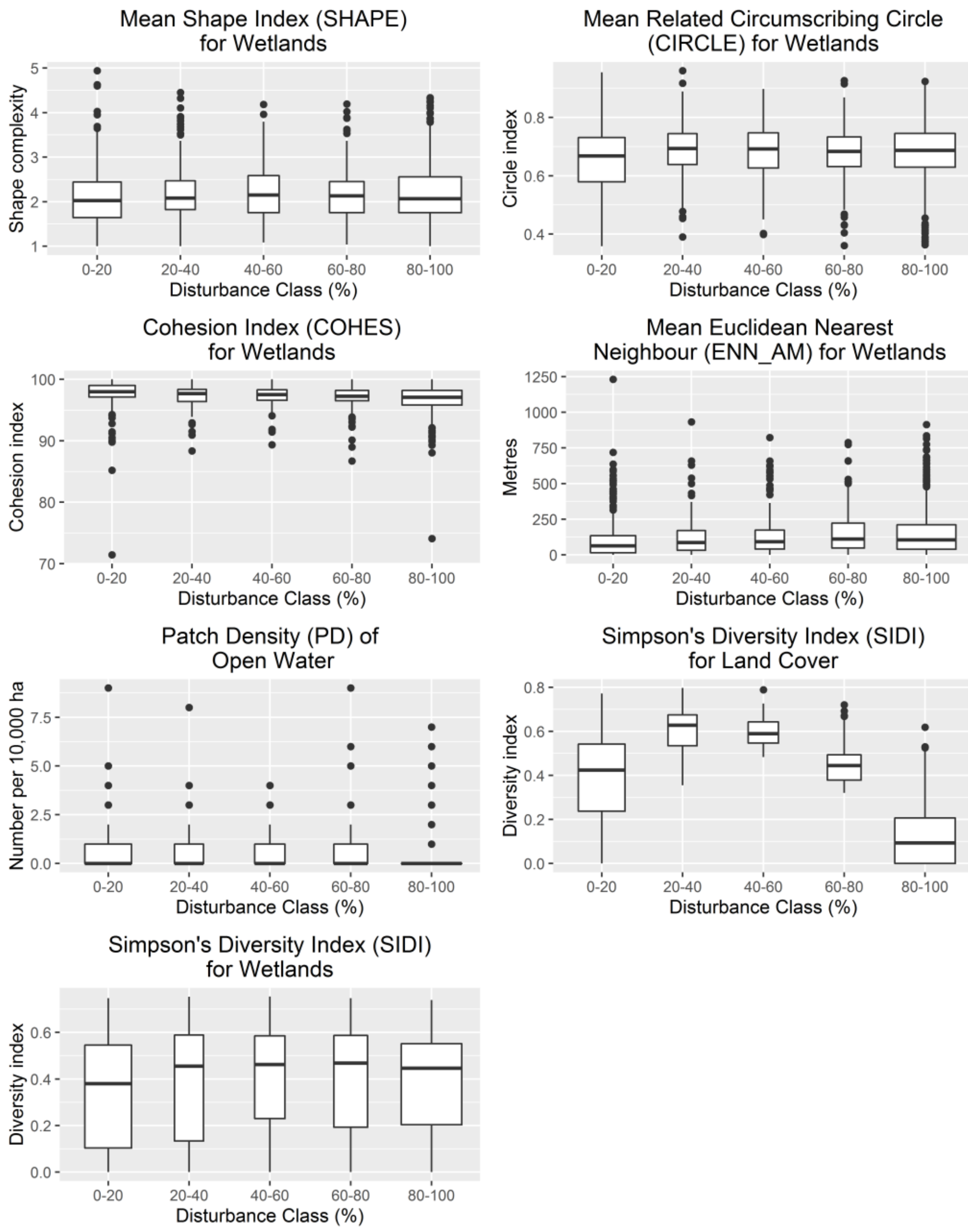

Figure 2.4: Metric distributions by disturbance intervals for the Central inventory sites. Boxplot widths are proportional to the number of observations in the group. Horizontal line is the median. Lower and upper edges of the boxes indicate 25^{th} and 75^{th} quartiles respectively. Whiskers extend to the extreme values within 1.5 times the inner quartile range (75^{th} percentile – 25^{th} percentile). Points are outliers.

The distribution of metric values for the Central inventory exhibit little visible difference between disturbance levels except for SIDI_LAND, which has a distinct peak at moderate disturbance levels (Figure 2.4). Despite the lack of visual variation in metric values across disturbance classes, quantitative comparisons with the Kruskal-Wallis tests for all metrics were significant ($p < 0.05$). However, H for SIDI_LAND was an order of magnitude greater than the other metrics (Table 2.6), which agrees with the boxplot observation that diversity of land cover varies more dramatically between disturbance levels relative to measures of wetland aggregation and shape complexity.

Table 2.6: Kruskal-Wallis test results for the significance of differences between metric values by disturbance intervals in the Central inventory (Central-All). *, ** and *** refer to significance levels $p < 0.05$, $p < 0.01$, and $p < 0.001$ respectively. Listed metrics are the response variables predicted by disturbance.

Type	Metric	H	p	Significance
Shape	SHAPE_WET	9.701	0.046	*
	CIRCLE_WET	27.611	<0.001	***
Aggregation	COHES_WET	105.207	<0.001	***
	ENN_WET	52.595	<0.001	***
	PD_OW	91.484	<0.001	***
Diversity	SIDI_LAND	1221.817	<0.001	***
	SIDI_WET	12.675	0.013	*

Having established statistically significant differences among metric values by disturbance level, pairwise comparisons between disturbance levels for each metric were calculated. All pairwise comparisons, except 20-40% vs. 40-60%, had at least one metric that differed significantly between disturbance levels (Table 2.7). The pairwise comparisons involving the least disturbed (0-20%) landscapes contrast with the other comparisons because most metrics in this disturbance class were significant (Table 2.8, first column). A similar pattern is also apparent for highly disturbed landscapes albeit with fewer metrics (Table 2.8, bottom row). This suggests that undisturbed and highly disturbed landscapes are distinguished from each other and intermediate disturbance landscapes in terms of wetland shape, aggregation, and diversity. Conversely, intermediate disturbance landscapes are only distinguishable in terms of land-cover diversity. Of the metric values that showed significant differences with disturbance levels, only SIDI_LAND was significant for all comparisons.

Table 2.7: Pairwise comparisons of disturbance classes in the Central inventory (Central-All) using the Dunn's test. *, ** and *** refer to significance levels $p < 0.05$, $p < 0.01$, and $p < 0.001$ respectively. Grey cells indicate redundant comparisons.

Disturbance Class (%)	0-20	20-40	40-60	60-80

20-40	CIRCLE_WET*** COHES_WET*** ENN_WET* PD_OW** SIDI_LAND***			
40-60	SHAPE_WET* CIRCLE_WET** COHES_WET*** ENN_WET** PD_OW** SIDI_LAND*** SIDI_WET*	none		
60-80	COHES_WET*** ENN_WET*** PD_OW*** SIDI_LAND* SIDI_WET*	SIDI_LAND***	SIDI_LAND***	
80-100	CIRCLE_WET*** COHES_WET*** ENN_WET*** PD_OW*** SIDI_LAND***	COHES_WET** PD_OW* SIDI_LAND***	COHES_WET** PD_OW** SIDI_LAND***	SIDI_LAND***

Like the Central inventory, many of the metric distributions did not visibly differ between disturbance levels for the Southern inventory (Figure 2.5). Significant Kruskal-Wallis results were found for 4 of the 6 Southern representative metrics though only 2 of these were at $p < 0.001$ (Table 2.8). Also similar to the Central inventory, H for SIDI_LAND is an order of magnitude greater than the other metrics, indicating a substantially higher amount of variation between disturbance classes (Table 2.8). The Dunn tests for the Southern inventory were similar to those of the Central inventory in that wetland and land cover patterns of the least disturbed (0-20%) and most disturbed (80-100%) landscapes were distinguished from each other and intermediate disturbance landscapes (Table 2.10, first column, and bottom row). However, the differences were less pronounced relative to the Central inventory since there were never more than 3 metrics with significant results per comparison (Table 2.10). Furthermore, the significance of the comparisons was never better than $p < 0.01$ for any metric other than SIDI_LAND, unlike in the Central inventory where most of the comparisons had $p < 0.001$.

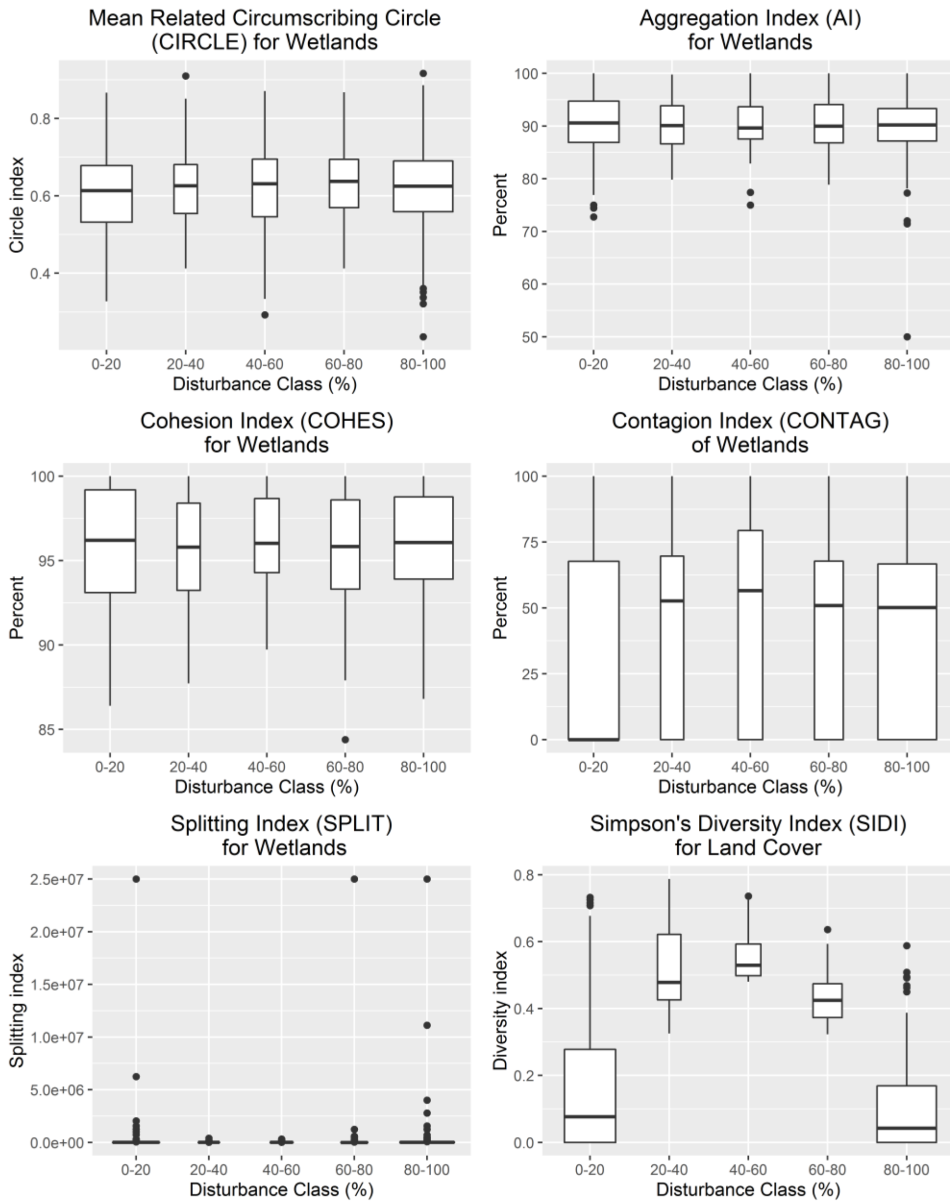

Figure 2.5: Metric distributions over disturbance intervals for the Southern inventory (Southern-All). Boxplot widths are proportional to the number of observations in the group. Horizontal line is the median. Lower and upper edges of the boxes indicate 25th and 75th quartiles respectively. Whiskers extend to the extreme values within 1.5 times the inner quartile range (75th percentile – 25th percentile). Points are outliers.

Table 2.8: Kruskal-Wallis test results for the significance of differences between metric values by disturbance intervals in the Southern inventory (Southern-All). *, ** and *** refer to

significance levels $p < 0.05$, $p < 0.01$, and $p < 0.001$ respectively. Listed metrics are the response variables predicted by disturbance.

Type	Metric	*H*	*p*	Significance
Shape	CIRCLE_WET	10.503	0.033	*
Aggregation	AI_WET	2.538	0.638	
	COHES_WET	2.039	0.729	
	CONTAG_WET	11.143	0.025	*
	SPLIT_WET	27.923	<0.001	***
Diversity	SIDI_LAND	742.237	<0.001	***

Table 2.9: Pairwise comparisons of disturbance classes in the Southern inventory using the Dunn's test. *, ** and *** refer to significance levels $p < 0.05$, $p < 0.01$, and $p < 0.001$ respectively. Gray boxes indicate redundant comparisons.

Disturbance Class (%)	**0-20**	**20-40**	**40-60**	**60-80**
20-40	SIDI_LAND***			
40-60	CONTAG_WET* SIDI_LAND***	none		
60-80	CIRCLE_WET* SIDI_LAND***	none	SIDI_LAND**	
80-100	SPLIT_WET** SIDI_WET***	SPLIT_WET* SIDI_LAND***	CONTAG_WET* SPLIT_WET** SIDI_LAND***	SPLIT_WET** SIDI_LAND***

3.3 – Relating Metrics to Disturbance Between Natural Regions

To determine if the influence of disturbance on landscape metric values holds across Natural Regions, a comparison of the metric distributions between Natural Regions was performed. Of the 35 comparisons for the Central inventory (7 metrics at 5 disturbance levels), 14 were significantly different with more than half of these at $p < 0.001$ (Figure 2.6, Table 2.10). The significant differences between Natural Regions were more prominent at higher disturbance levels with 9 of the 14 significant differences occurring in the top two disturbance classes. Patch density of open water (PD_OW) was significantly higher ($p < 0.001$) in Parkland for the top three disturbance classes (Figure 2.6).

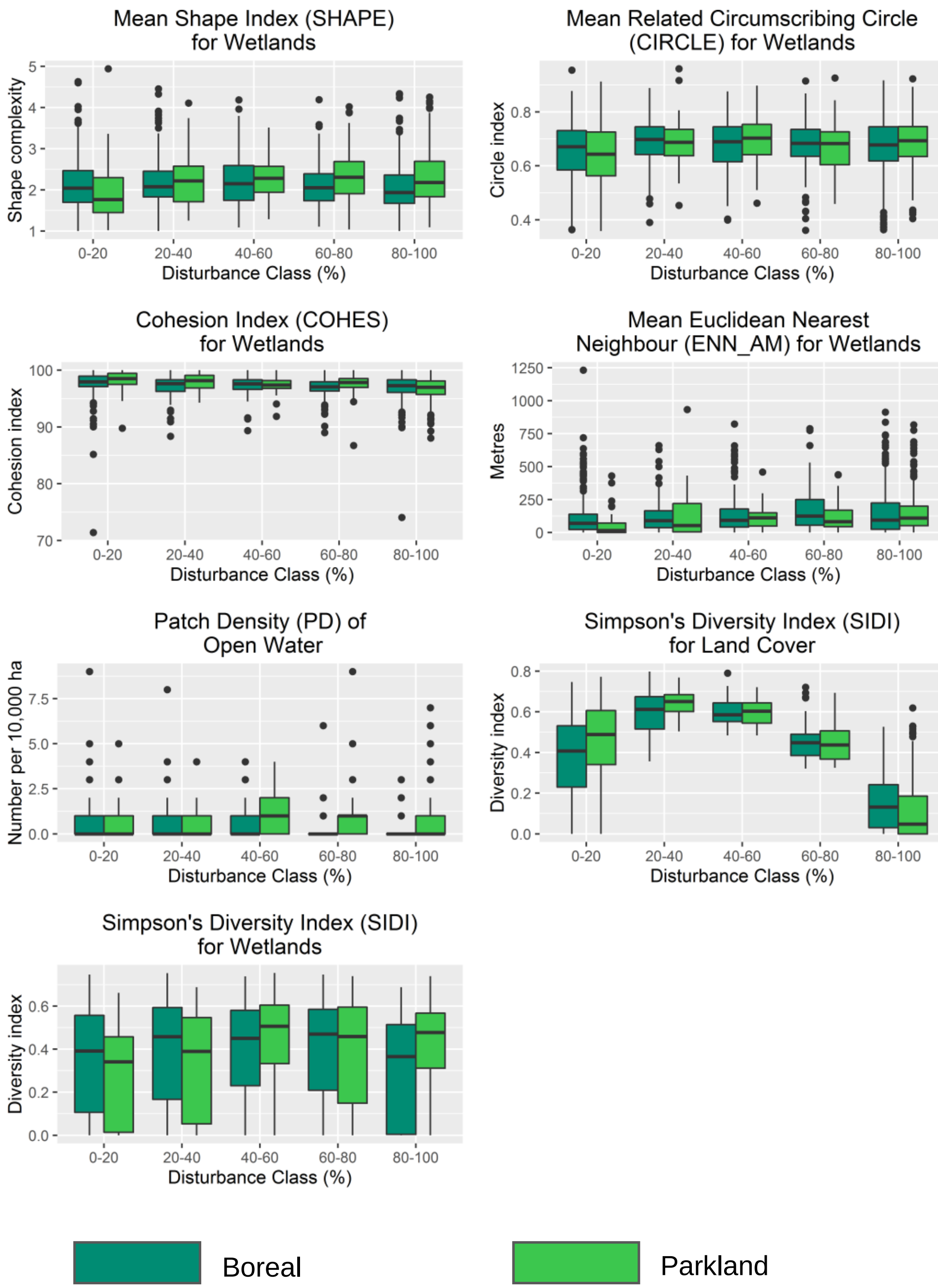

Figure 2.6: Comparison of metric values across disturbance between Natural Regions in the Central wetland inventory.

Table 2.10: Significance levels (p-values) of the Kruskal-Wallis comparisons of metric distributions between Natural Regions (Boreal and Parkland) at corresponding disturbance levels

in the Central inventory. *, ** and *** refer to significance levels $p < 0.05$, $p < 0.01$, and $p < 0.001$ respectively.

Metric	Disturbance (%) 0-20	20-40	40-60	60-80	80-100
SHAPE_WET	0.024*	0.887	0.656	0.003**	<0.001***
CIRCLE_WET	0.409	0.651	0.338	0.566	0.067
COHES_WET	0.05	0.063	0.724	<0.001***	0.01*
ENN_WET	<0.001***	0.321	0.917	0.034*	0.112
PD_OW	0.228	0.328	<0.001***	<0.001***	<0.001***
SIDI_LAND	0.013*	0.05	0.616	0.722	<0.001***
SIDI_WET	0.041*	0.248	0.293	0.826	<0.001***

Comparisons of the metric distributions between Natural Regions in the Southern inventory, indicated that there is little difference in metric values (Figure 2.7). Of the 30 comparisons made, only 6 had statistical significance ($p < 0.05$; Table 2.11). The most notable differences between Natural Regions were with land-cover diversity (SIDI_LAND), which was significantly higher in Parkland for the two lowest disturbance levels. The differences among land-cover diversity is expected because Grassland is dominated by grass in undisturbed areas whereas Parkland has a mix of grass, shrub, and forest (Appendix E).

Table 2.11: Significance levels (p-values) of the Kruskal-Wallis comparisons of metric distributions between Natural Regions (i.e. Grassland and Parkland) at corresponding disturbance levels in the Southern inventory. *, ** and *** refer to significance levels $p < 0.05$, $p < 0.01$, and $p < 0.001$ respectively.

Metric	Disturbance (%) 0-20	20-40	40-60	60-80	80-100
CIRCLE_WET	0.464	0.255	0.207	0.929	0.935
AI_WET	0.086	0.338	0.041*	0.912	0.905
COHESION_WET	0.358	0.446	0.503	0.254	0.762
CONTAG_WET	0.081	0.191	0.374	0.537	0.091
SPLIT_WET	0.017*	0.288	0.603	0.137	0.012*
SIDI_LAND	<0.001***	<0.001***	0.064	0.69	0.013*

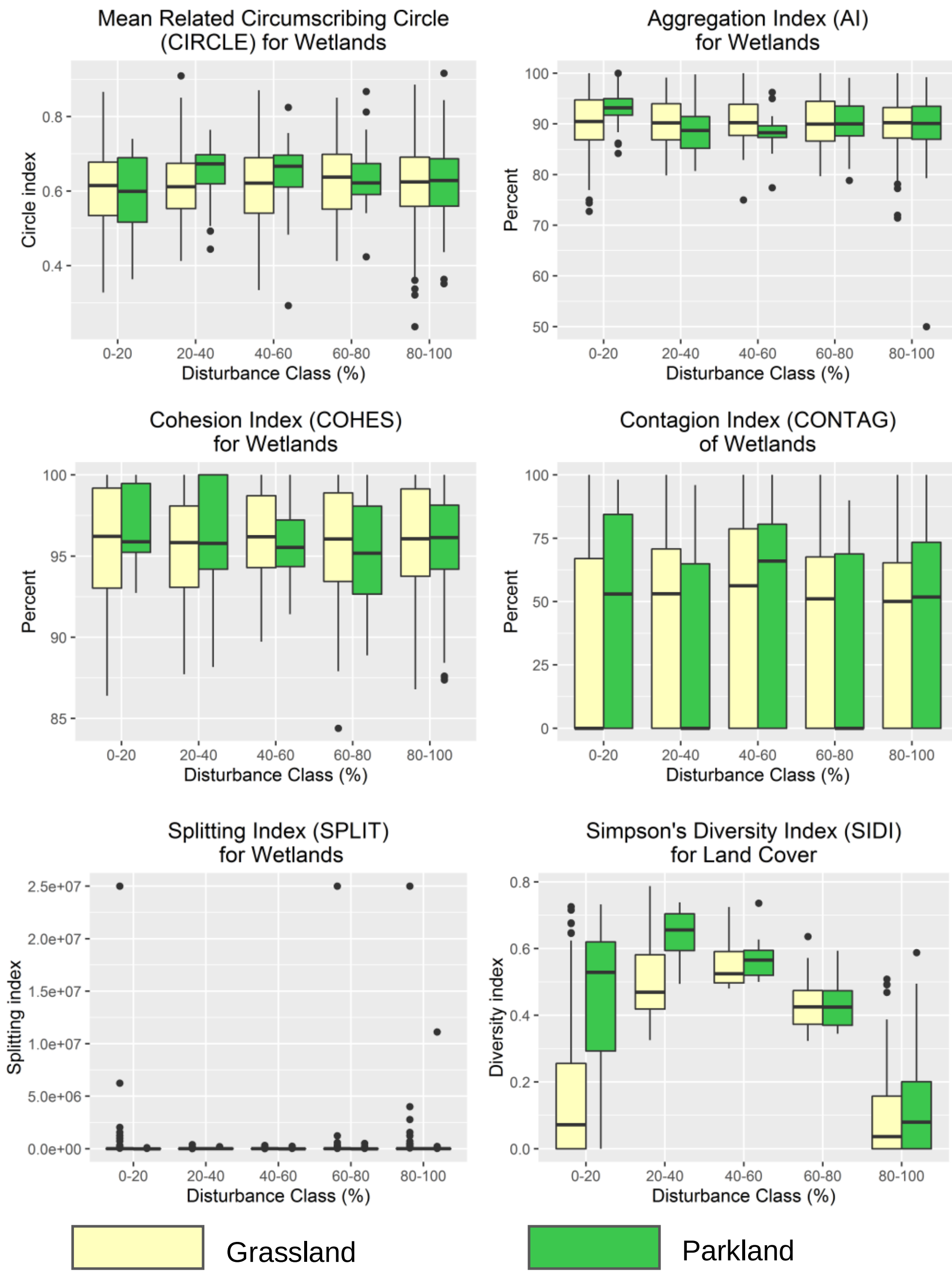

Figure 2.7: Comparison of metric values across disturbance between Natural Regions in the Southern wetland inventory

4 – Discussion

4.1 – Association of Disturbance Levels with Metric Values

Our study of land-cover pattern across a gradient of disturbance in wetland landscapes suggests that human disturbance significantly effects land-cover pattern. Metrics used to quantify land-cover pattern, for both Central and Southern inventories, were sensitive to the proportion of disturbance (i.e. agricultural and urban lands) in the landscape. Pairwise comparisons of land cover pattern between disturbance intervals revealed that undisturbed landscapes (i.e. 0-20%) and highly disturbed landscapes (i.e. 80-100%) were significantly different from each other, and from landscapes with intermediate disturbance (i.e 20-40%, 40-60%, or 60-80%). The fact that undisturbed and highly disturbed landscapes are significantly different from each other and intermediate disturbance landscapes is noteworthy because it suggests that a reference condition approach (Bailey et al., 2004; Hawkins et al., 2010) can be utilized to assess if reclaimed landscapes appear the same as natural ones. Typically, a reference condition approach is applied for ecosystems whereby the variability of indices of biotic integrity (IBI) in pristine ecosystems (i.e. minimal exposure to anthropogenic stressors) are empirically quantified, and used as a benchmark for assessing the health of other ecosystems (e.g. Hawkins et al, 2010; Karr, 1991; Kennar et al., 2006; Landres et al., 1999; Pardo et al., 2012; Reynoldson et al., 1997; Stoddard et al., 2006; Tonn et al., 2011). Since the results of this study show that wetland landscapes have different composition and configuration between disturbance intervals, I argue that the traditional reference condition approach (i.e. using pristine landscapes as the sole benchmark) can be modified so that different reference conditions are defined for each disturbance interval.

Approaches analogous to the reference condition have been applied to assess landscape pattern (rather than biotic integrity), where remotely-sensed imagery is used to quantify the differences between historical (pre-disturbance) and modern (i.e. post disturbance) forested landscapes (e.g. Abella & Denton, 2009; Hessburg et al., 2004; Keane et al., 2002; Keane et al., 2009). Although these studies used historical imagery of forest extent as a baseline, rather than biotic characteristics, they are similar to the reference condition approach used for bioassessment since samples of undisturbed locations are used to define benchmarks. Research using historical imagery as a baseline has been termed “historical range and variability” (HRV).

Wetland landscapes have received less attention than forested landscapes in HRV research. Exceptions are Liu & Cameron, (2001) and Li et al (2010) who observed changes in wetland shape complexity and aggregation with increasing levels of human disturbance. Our results corroborate these findings and provide 1) a methodology for the identification of landscape metrics for comparison between reference and reclamation sites in other study areas, and 2) a quantification of pattern for use by others for comparison or reclamation along a gradient of disturbance in multiple Natural Regions, and with different data quality, for the PPR.

HRV may not always be feasible for wetland reclamation because it may not be possible to return a landscape to its exact original state. For example, climate models predict that Grassland and Parkland regions will shift northward into the Boreal region over the next century due to climate change (Schneider, 2013). Climate has a major impact on wetland hydrology (Dawson et al., 2003), and soil chemistry (Davidson & Janssens, 2006), so wetland reclamation planning needs to account for climate change to be sustainable. I argue that this study's method of sampling landscapes existing in climates that will move into the Boreal region is more informative for wetland reclamation planning than using HRV of the Boreal to set baselines.

An essential step of a reference condition approach is to answer why the test sites being compared against reference sites are impaired (Bowman & Somers, 2005). For the Central inventory, 4 metrics were significantly different for each of the pairwise comparisons between undisturbed landscapes and all other disturbance intervals (Table 2.7, column 1). Three of these metrics, cohesion index for wetlands (COHES_WET), patch density of open water wetlands (PD_OW), and area-weighted mean Euclidean nearest neighbour (ENN_WET) were aggregation metrics. COHES_WET is a measure of the physical connectivity of wetlands, and PD_OW is measure of the number of wetland patches per unit area. Both of these measures decreased with the increasing proportion of non-natural land cover in a landscape, and this was expected because wetland area is lost when agricultural development occurs in the PPR (Higgins, 1977; Sugden & Beyersbergen, 1984). Overall wetland-area loss in a landscape is not a perfect predictor of wetland aggregation however, as indicated by the fact that the identified aggregation metrics in this study were not highly correlated with percent-wetland in the landscape. There was substantial overlap in the range of wetland aggregation metric values between undisturbed landscapes and disturbed landscapes. This is likely due to the fact that wetlands often occupied only a small proportion of the sample landscapes for all disturbance intervals, while other natural land covers occupy greater proportions of land. Of the natural land covers, grass is dominant in the Grassland, grass and shrub are dominant in Parkland, and forest is dominant in Boreal (Appendix E). Despite the overlap of wetland aggregation across disturbance intervals, the Kruskal-Wallis and Dunn's tests indicate that wetland aggregation significantly varies with disturbance.

Significant differences were also identified for Simpson's diversity index for land cover (SIDI_LAND) for all but one of the pairwise comparisons between disturbance intervals (Table 2.7). SIDI_LAND is lowest when landscapes are dominated by a single land cover, and is highest when landscapes contain all possible patch types in equal proportions (McGarigal, 2014). SIDI_LAND exhibited a parabolic pattern, for both Central and Southern inventories, where the metric values are lowest for the least and most disturbed landscapes and higher for the intermediate classes. This is an expected occurrence because the least and most disturbed landscapes are typically dominated by a small number of land covers (i.e. agriculture for areas of high disturbance, or one of natural grass, shrub, or forest in areas of low disturbance). The parabolic shape of the SIDI_LAND distributions across disturbance intervals is partially the result of the reclassification of the AAFC crop inventory data. The reclassification consolidated 22 agricultural classes into 1 with the assumption that the different crop types would have similar affects to wetlands. This likely had the effect that SIDI_LAND was lower for highly disturbed areas simply because the number of crop types was masked. However, it is expected that highly disturbed landscapes would still be less diverse than intermediate-disturbance landscapes because they are less likely to have substantial presence of natural land covers. Undisturbed landscapes were less affected by the reclassification of the AAFC crop inventory because the only land cover class consolidation was for the 3 forest types, which were merged together. This likely lowered the land-cover diversity for the Boreal sample landscapes, where forested areas are dominant (Appendix E), while Grassland and Parkland sample landscapes likely experience minimal effects because of the relatively small amount of forest (Appendix E).

4.2 – Effect of Natural Regions on Metric Selection and Values

Comparison of metric selection across Natural Regions suggests that the same metrics of wetland configuration in one Natural Region are not necessarily appropriate to be used as targets for reclamation in other regions. The selected metrics for configuration were not consistent between Natural Region subsets within the same inventory (i.e. Central Boreal vs. Central Parkland and Southern Parkland vs. Southern Grassland). Relaxing the selection procedure still resulted in few common representatives between Natural Region subsets within the same inventory (4 of 9 for Central and 2 of 7 for Southern; Appendix B). The implication of this is that a single set of metrics cannot be used over large spatial extents encompassing different Natural Regions. This echoes ecosystem-level research where indicators of ecosystem integrity vary among ecosystems with different community compositions (Carignan & Villard, 2002). Despite the differences in specific representative metrics between subsets, aggregation metrics were frequently identified, suggesting that aggregation is generally the most informative of all metric types in quantifying wetland configuration.

Comparison of Natural Regions at corresponding disturbance levels suggests that there are more similarities between Grassland and Parkland than there are between Parkland and Boreal. More than twice as many significant differences were found comparing Parkland to Boreal (14 of 35 comparisons; Table 2.10) than Grassland to Parkland (6 of 30 comparisons; Table 2.11). This is likely due to Parkland and Grassland having a more similar composition than Parkland and Boreal. Boreal is dominated by forested land and permanent organic wetlands, while Parkland and Grassland contain minimal forest cover and the wetlands are non-permanent mineral prairie potholes.

4.3 – Method for Metric Selection

Metric reduction is necessary because some metrics have similar interpretive value or they are empirically correlated (Cushman et al., 2008). Various combinations of correlation grouping, regression with habitat proportion, and PCA have been frequently used in landscape ecology research to select a manageable set of landscape metrics (Herzog et al., 2001; Mairota et al., 2013; McGarigal & McComb, 1995; Plexida et al., 2014; Riitters et al., 1995). These techniques were applied to specifically quantify wetland composition and configuration rather than overall landscape pattern. In doing so, wetland aggregation was identified as the dominant axis along which to quantify wetland configuration. The prominence of aggregation in the final landscape metric selection was in large part due to 1) redundancy of representing shape and diversity/evenness using multiple metrics and 2) the association of many area/edge metrics with percent wetland, which were removed. The presented method of removing redundant metrics through correlation grouping is standard practice. However, the method of removing metrics associated with percent-wetland in this study differs with other studies, which instead retained residual metrics representing the variation of configuration independent of land-cover proportion (Cushman et al., 2008; Mairota et al., 2013; McGarigal & McComb, 1995).

Residual metrics are the residuals of a regression between land-cover proportion and a given configuration metric. A residual metric does not retain the same scale as the original metric, and is dependent on knowledge of habitat proportion to be interpreted. Residual metrics can be useful because they allow for landscape pattern to be described in more detail with a greater number of axes (Appendix F). However, this study demonstrates that many regular (i.e. non-residual) shape, aggregation, and diversity metrics vary independently of land-cover proportion. These metrics can be used in conjunction with simple measures of wetland proportion for simulating landscapes to guide wetland reclamation. It was therefore more prudent to only retain metrics not related to wetland proportion, since proportion would be used as a design parameter.

4.4 – Ecologic and Hydrologic Relevance

Connectivity between wetland patches and their surrounding landscapes occurs ecologically (Galatowitsch & van der Valk, 1996; Haig et al., 1998) and hydrologically (Cohen et al., 2016; Wilcox et al., 2011), and thus landscape metrics quantifying wetland and land-cover configuration are important considerations when reclaiming landscapes with targeted ecologic and hydrologic functions. Many restored and constructed wetlands have been unable to reach natural levels of biodiversity, and this has been partly attributed to a lack of dispersal (Galatowitsch & van der Valk, 1996). Less aggregation of wetland patches likely inhibits dispersal because of greater distance between patches. Aggregation of wetland patches also affects obligate bird species, which move between individual wetlands (Haig et al., 1998). Landscapes where wetland patches are more aggregated (i.e. higher patch density) have higher duck populations, and duck predator species (Stephens et al., 2005).

Diversity of wetland types in a landscape also has ecological implications. Temporary and seasonal wetlands benefit some amphibian species who use these wetlands as safe breeding grounds whereas wetlands with longer hydroperiods are more conducive to predator fish that consume the amphibians (Babbitt, 2005). It is therefore beneficial for landscapes to contain wetlands with a diversity of hydroperiods to maximize biodiversity. Land-cover diversity and interspersion (an aspect of aggregation) also influence butterfly dispersal; the probability of occurrence of *B. titania* was increased with the presence of forest cover in between patches of wetland habitat (Cozzi et al., 2008).

The spatial configuration of wetland patches is also influential on landscape hydrology. Wetlands exists along a continuum of hydrologic connectivity and the degree of connectivity influences wetlands' hydrologic function (Leibowitz, 2003). A higher level of wetland aggregation allows for greater surface water connectivity between wetlands, and wetlands with persistent surface connectivity provide storage during times of peak flow (Cohen et al., 2016). Conversely, wetlands without only intermittent surface connectivity (i.e. low aggregation PPR wetland landscapes) control base flow, and limit peak flow and recession rates (Cohen et al., 2016). Hydrologic connectivity also has a direct impact on wetland nutrient and pollutant retention. Less connected wetlands have superior retention because there is less surface water flow to remove nutrients and pollutants from wetlands (Marton et al., 2015). Wetlands in the PPR therefore provide many hydrological benefits, despite their lack of surface connectivity.

4.5 – Limitations

Comparing the selected metrics for the two wetland inventories within the same natural region, which have similar climate, landscape composition, and wetland distribution, highlight the sensitivity of landscape metric selection to data quality. The Parkland subsets of the two inventories (i.e. Central Parkland and Southern Parkland) identified only one common metric (COHES_WET), and significant differences were found in the distributions of COHES_WET between the two inventories for the three highest disturbance levels (Appendix G).

Another aspect of data quality that likely influenced metric selection and values is the classification system of categorical wetland and land-cover data. Most of the calculated metrics quantify aspects of wetland configuration and diversity but, since wetlands do not exist as biogeographic islands (Herrmann et al., 2005; Matthews et al., 2009), we used the AAFC crop inventory to quantify diversity and evenness of other land covers. The AAFC crop inventory was the best available land cover data that covered the entire study area but there were a number of limitations with this approach. The AAFC crop inventory was derived using a different methodology than either of the wetland inventories at a coarser resolution (56 m) it was created for a different time period so it was deemed inappropriate to use these data to calculate area, shape and aggregation measures. Furthermore, the main intent of the AAFC crop inventory was to accurately delineate crop types while the other land covers were not as thoroughly assessed for accuracy. A result of the focus of the AAFC crop inventory was that wetland areas identified in the crop inventory did not always align well with the wetland areas of the wetland inventories. While the existing crop inventory data set is sufficient to quantify land cover class diversity/evenness, it would have been more ideal to have a single high thematic and spatial resolution land-use and land-cover that covered the study area and identified permanence classes of wetlands. Data with this level of comprehensive spatial and attribute coverage do not exist. The necessity of harmonized data is consistent with the findings of Lausch & Herzog (2002), who observed that metrics values can differ simply due to the use of data sources created using differing methods.

Positive spatial autocorrelation (i.e. observations made in nearby spatial locations are not independent from each other; Dormann et al., 2007) can confound statistical tests that assume independence of observations. When observations are not fully independent, the effective sample size is less than the total number of observations, and the likelihood of making a Type I error (i.e. false positive) increases (Dale & Fortin, 2002; Overmars, De Koning, & Veldkamp, 2003). Though the presented research did not formally quantify spatial autocorrelation, a high degree of spatial structure in the sample sites is apparent, with highly-disturbed landscapes clustered in the west of the study area and the less disturbed landscapes clustered in the east (Figure 2.3). In the presented study, the Bonferroni adjustment was applied to the pairwise Dunn tests to counteract the effect of spatial autocorrelation. The Bonferroni adjustment divides the critical value by the number of comparisons made to reduce the likelihood of identifying false positives when comparing many groups (Dunn, 1964; Renard, Demougeot, & Froidevaus, 2005). There are other methods for reducing the likelihood of false positives, though the Bonferroni is often considered among the most stringent (Feise, 2002), and therefore could reliably ensure that significant results were actually significant.

5 – Conclusions

The ecologic, hydrologic, and economic importance of wetlands underscores the need for sustainable reclamation strategies. One essential consideration for developing successful long-term wetland reclamation strategies is the need to quantify composition and configuration at the landscape scale using landscape metrics. A key finding of this research is that wetland configuration in low and high disturbance landscapes significantly differ from each other and intermediate disturbance landscapes. The links between wetland landscape configuration, and hydrologic and ecologic function mean that reclaimed landscapes will require similar spatial configurations to natural, undisturbed landscapes for comparable functionality to be reached. Aggregation is the primary means of measuring wetland landscape configuration independent of composition, as indicated by the predominance of aggregation representative metrics. Wetlands are more aggregated in low-disturbance landscapes, which supports the findings of previous research.

Many measures of wetland configuration vary predictably with the percent of wetland area in the landscapes, echoing the findings of past research that show that configuration is affected by composition. Embracing or removing the association between configuration and composition is conceptual choice dependent on the intended application. An application of this study is to use the values of selected configuration and diversity metrics for parameterizing the design or simulation of reclaimed wetland landscapes. Using residual metrics to quantify landscape pattern can be beneficial because they retain a greater number of axes to quantify landscape pattern (relative to the number of representatives in this study), but their abstract interpretation make landscape design difficult. The wetland configuration metrics identified in the presented research are independent of landscape composition and can be used in conjunction with simple measures of wetland proportion to design landscapes that mimic the structure of natural landscapes. The effect of Natural Regions on metric values indicates that baseline conditions for landscape design will vary depending on spatial location.

Caution should be taken when using disparate datasets to quantify wetland landscape pattern because of the impact on metric selection and metric values. In the context of landscape-scale wetland reclamation, data would have to meet three basic requirements for it to be of sufficient quality. The first is a spatial extent that covers a representative proportion of a relatively homogeneous ecological, biophysical, or climate region of interest (e.g. Natural Region). The second is consistent spatial, temporal, and thematic resolution derived from a single methodology for land use, land cover, and wetland classification across the study area to ensure that metrics are comparable. The last is high accuracy in wetland boundary delineation and classification to ensure that the configuration and composition metrics accurately reflect the real conditions present in the landscapes.

Chapter 3 – Context and Future Directions

1 – Context Within Landscape Ecology

Landscape ecology is an interdisciplinary scientific field with the goals of 1) identifying the reciprocal linkages between site-level ecological processes with landscape characteristics (Risser, 1999), and 2) communicating these linkages so that they can be used as a basis for land-use allocation in landscape planning (Ahern, 1999). However, integrating the concepts of ecological knowledge into landscape planning decisions remains a challenge due to the difficulty in relating site processes with landscape characteristics (i.e. Goal 2 cannot be completed without Goal 1; Vos et al., 2009; Wu & Hobbs, 2002). This challenge is worth addressing because successful integration of ecological knowledge into landscape planning can result in ecologically sustainable landscapes that maintain ecological services and are resilient to change over time (Opdam et al., 2006).

The research presented in Chapter 2 aligns with the landscape characteristics side of Goal 1, whereby variable reduction procedures were applied to metrics quantifying the configuration and diversity of wetlands. The identified metrics from the reduction procedures were then used to assess the degree to which wetland and land-cover composition and configuration change with different levels of anthropogenic disturbance, quantified by the extent of agricultural and urban areas, in wetland landscapes. Several key findings from this research contribute to the general knowledge of landscape ecology.

Wetland configuration and composition is significantly different in low-disturbance landscapes compared to landscapes at higher disturbance levels. While there is rich set of literature comparing landscape composition and configuration to anthropogenic disturbance (e.g. Griffith et al., 2003; Krummel et al., 1987; Luck & Wu, 2002; Miller et al., 1997; O'Neill et al., 1988), relatively little research has been conducted specifically focusing on wetland pattern with human disturbance. A notable exception to this trend is Li et al. (2010), who found small differences in wetland configuration with increasing levels of urbanization. However, Chapter 2 is differentiated from Li et al. (2010) in that it is a larger spatial extent, comprising different natural regions. Despite the differences in spatial extent and natural regions, both studies demonstrate that wetland aggregation decreases (i.e. the patches are more fragmented) with increasing proportions of non-natural land cover.

Results corroborate research in other fields and geographic locations that many metrics are redundant and are highly correlated with each other or land-cover proportion (e.g. Cushman et al., 2008; Herzog et al., 2001; McGarigal & McComb, 1995; Riitters et al., 1995). Residual metrics (the residuals of a regression model between land cover proportion and a given configuration metric) are frequently used to avoid confounding composition (i.e. percent-wetland Chapter 2) with configuration. However, there would be difficulty in applying the residual metrics for an application like designing landscapes with certain configuration parameters because the interpretation of residual metrics (departure from expected configuration as a given composition level) is more abstract that a standard landscape metric. Chapter 2 demonstrates that there are multiple configuration metrics that are independent of composition that can be used as parameters in landscape design.

The spatial location of sample sites also affected metric selection since representative metrics for Natural Region subsets within the same wetland inventory differed. The implications of these findings is that a single set of representative metrics cannot be chosen when using multiple datasets that cover large spatial extents. Though previous research has sampled over similarly large spatial extents (Cushman et al., 2008; Riitters et al., 1995), Chapter Two is the first time, to the best of my knowledge, that the effects of locating sample sites in areas of differing data quality and biophysical characteristics on metric selection have been explicitly explored. These findings are informative for reclamation planning because they indicate that configuration of reclaimed landscapes will vary depending on the location of the reclaimed site.

Representative metrics selected from variable reduction techniques are also affected by data quality. The wetland metrics were calculated from two different wetland inventories with non-overlapping spatial extents. The wetland inventories were created and quality assessed using different methods, and were visibly different in appearance (Figure 3.1). The Central inventory contains a greater diversity of different wetland classes and larger polygons relative to the Southern inventory. Some differences between inventories are likely due the differing spatial extents (Boreal and Parkland covered by Central; Parkland and Grassland covered by Southern), but the differences at the neighbouring boundaries of the two inventories suggest that methodological differences are also a factor. As such, sample sites within different inventories were analyzed separately, and few of the identified representative metrics were common between inventories.

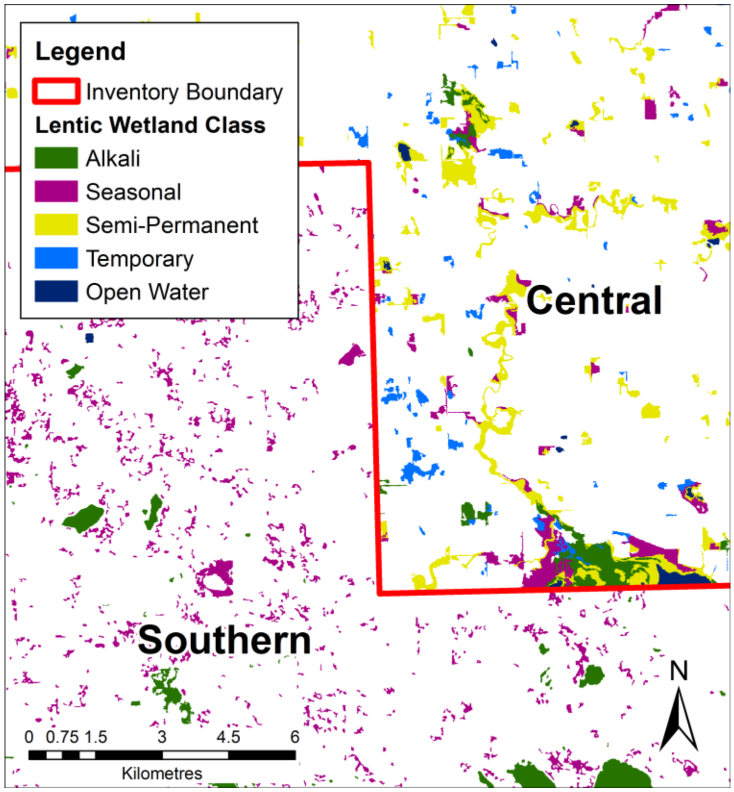

Figure 3.1: A section of the boundary between the wetland inventories (Central vs. Southern) used in Chapter 2. Though both inventories use the same classification system, the differences in wetland delineation and classification are clearly visible.

2 – Future Directions

While Chapter 2 makes new contributions to the field of landscape ecology, it is limited by the fact that it only addresses half of the landscape ecology Goal 1 (i.e. quantifying landscape pattern). Future research needs to take the identified measures of wetland configuration and composition and relate them to site-level ecological and environmental characteristics to complete the mandate of Goal 1 (i.e. relating site-level ecological processes with landscape characteristics). Perhaps even more challenging will be translating the identified linkages into a scientifically-sound template for landscape-scale wetland reclamation. With these limitations in mind, the following section will 1) describe preliminary research undertaken to relate wetland landscape characteristics with site-level environmental and ecological variables, and 2) propose a framework for a sustainable wetland reclamation strategy that aligns with recent legislation changes in the province of Alberta.

2.1 – Comparing Site-Level Variables with Landscape Composition and Configuration

Landscape ecology is founded on the notion that landscape conditions can at least partially explain variation in site-level conditions. This theory has been tested for wetland habitats through studies that investigate the association between landscape composition and wetland condition (e.g. Lopez et al., 2002; Mack, 2006; Matthews et al., 2009), though the influence of landscape configuration has not been examined as thoroughly (with exceptions such as Vos & Stumpel [1995], Brown & Dinsmore [1986], and Pérez-García et al. [2014], all of whom found wetland patch isolation to have a negative relationship with species richness and/or abundance). The relative lack of consideration for the effect of landscape configuration on wetland condition is a concern because landscape structure cannot be fully described by compositional measures alone. The spatial configuration of wetland patches influences wetland condition through its effects on wetland hydrology (e.g. Leibowitz & Vining, 2003), and the dispersal (e.g. Cozzi et al., 2008; Haig et al., 1998) and survival of biota (e.g. Stephens et al., 2005).

In addition to the above-cited studies, Kraft et al. (unpublished) have examined the relationships between landscape characteristics and indicators of biological integrity (IBI) for non-permanent marshes in the Prairie Pothole Region (PPR) of Alberta (i.e. the region where most of the Chapter 2 sample sites were situated). The study found weak, significant associations between land-cover composition and observations of site-level vegetation community composition and environmental variables. The authors noted that it would likely be important to consider land-cover configuration, in addition to composition, for complex agricultural landscapes like the PPR.

To investigate the associations of site-level observations with landscape characteristics, the representative configuration and diversity metrics from Chapter 2, calculated for 500 m buffer areas around the 48 sites visited by Kraft et al. (unpublished) were plotted against site-level observations (Table 3.1). In addition to the plots, a series of simple regressions (linear, quadratic, exponential) were run to quantify the statistical associations between site observations and landscape variables. Since most sites were situated in the Southern inventory (41 of 48), only the Southern inventory metrics from Chapter 2 were used as predictor variables in the regressions.

A visual interpretation of the plots indicates weak associations at best for all comparisons, and correspondingly weak r^2 values (i.e., < 0.2) from the regressions support the visual observation. Conversely, when plotting relative area proportion measures (e.g. percent forest in the landscape), some moderate to strong associations were found. For example, as one would expect, percent forest had a strong positive association (r^2 = 0.81) with forest dwelling bird species using a quadratic regression model. Percent shrub land and percent forest, which have high litter inputs, were also both moderately associated (r^2 = 0.68 and 0.66 respectively) with soil loss on ignition using quadratic regression models. However, despite these relatively strong associations, most landscape composition measures were weakly associated with the site variables, like the measures of wetland configuration identified in Chapter 2.

Table 3.1: Selected site-level variables measured for 48 wetlands in Spring 2014. Environmental and vegetation variables are a subset of the ones used by Kraft et al. (unpublished). Observations for the bird variables were collected during the same campaign but not used by Kraft et al. (unpublished).

Category	Variable
Environmental (Soil)	Average loss on ignition
	Potassium
Environmental (Water)	Water Conductivity
	Amplitude
	Max Depth
	Total Suspended Solids
Vegetation	Average coefficient of conservatism
	Floristic quality index
	Native species richness
	Exotic species richness
	Wetland-obligate species richness
Birds (Habitat)	Near water
	Forest dweller
	Field scrub
Birds (Groups)	Waterfowl
	Passerine

Due to the complexity of natural ecosystems, simple relationships rarely yield high r-squared values, so these weak associations are still considered ecologically meaningful. Multiple regressions can accommodate added complexity by allowing the use of more than one predictor variable. However, multiple regressions require selection of appropriate predictor variables; strong but non-significant associations can be found by simply adding more predictor variables to a model. More importantly, multiple regression only examines one response variable at a time, while it may be more desirable to determine if samples with generally similar landscape characteristics also tend to have similar site characteristics. Mantel tests are commonly used in ecology studies to answer these types of questions by correlating two pairwise dissimilarity matrices (derived from either a single vector or entire data tables; Legendre et al., 2015; McCune & Grace, 2002).

In addition to selecting the appropriate statistical tests, inclusion of terrain analysis (i.e. topography) in the quantification of landscape characteristics will likely help relate landscape characteristics to site properties (Dorner et al., 2002). For example, if two wetland patches are located at an equal distance but different direction from the nearest farm, the surface runoff from the landscape will likely not affect wetlands in the same way if the topographies (e.g. slopes and aspects) of the landscapes differ. Furthermore, it has been demonstrated that wetland locations and inundation frequency can be predicted using topography metrics (Lang et al., 2013), suggesting that topography plays a prominent role in shaping the spatial arrangement of patches quantified by landscape metrics. Regardless of the statistical methods chosen for comparing landscape and site characteristics, it is likely that more meaningful associations will be found between site and landscape characteristics with incorporation of terrain analysis.

2.2 – Informing Wetland Reclamation Policies

The research presented in Chapter 2, and the preceding section of the current chapter, were situated in the Province of Alberta with the intent to create wetland reclamation standards aligned with the Alberta Wetland Policy introduced in 2013 (Alberta Environment and Sustainable Resource Development, 2013). The Alberta Wetland Policy improves upon the preceding interim policy (Alberta Water Resources Commission, 1993) because it is applied to the entire province (rather than just settled areas), and provides a framework for assessing the relative value of wetlands. Relative wetland value is determined within Relative Wetland Value Assessment Units (RWVAU; geographic areas delineated based on climate and ecological similarity) to ensure that overall value within an assessment unit is maintained.

Currently, the Alberta Wetland Rapid Evaluation Tool (ABWRET) is used to place a monetary value on wetlands based on indicators broadly grouped into four categories: water quality, hydrologic function, biodiversity and ecological health, and human use (Government of Alberta, 2015). Sites are classified as A (highest value), B, C, or D (lowest value) based on the relative abundance of their functions within the RWVAU. This classification system is designed to discourage the development of high value (i.e. class A) wetlands by assigning them a higher replacement cost relative to lower value wetlands (Table 3.2). In scenarios where wetland loss is unavoidable, developers are expected to compensate for the loss by adding an equal amount of wetland value elsewhere within the RWVAU.

Table 3.2: Wetland Replacement Matrix (Hebben, 2013). Ratios are expressed as hectares of wetland.

		Value of Replacement Wetland			
		D	C	B	A
Value of Lost Wetland	A	8:1	4:1	2:1	1:1
	B	4:1	2:1	1:1	0.5:1
	C	2:1	1:1	0.5:1	0.25:1
	D	1:1	0.5:1	0.25:1	0.125:1

The shortcoming of ABWRET as an assessment tool for wetlands is that it only provides a quick estimate of relative wetland value to humans, but does not measure the ecological functions of wetlands, which would be necessary for determining if constructed wetlands successfully integrate with the natural landscape. This shortcoming can be addressed by using IBIs to measure the functionality of wetlands. Targets for constructed wetlands can be set using a reference condition approach (Bailey et al., 2004), where IBIs are measured for natural wetlands, and those measured values are used as benchmarks for constructed wetlands. To create self-sustaining wetlands that mimic the functionality of natural wetlands, landscape characteristics (composition, configuration, topography) will have to be considered for two reasons: 1) many restoration projects would be undertaken for large spatial extents encompassing many wetlands (e.g. Alberta Oil Sands extraction projects), and 2) the association between wetlands and their encompassing landscape means that achieving desired ranges of site-level IBIs would require landscape indicators to also be within desired ranges. As such, multi-scale criteria would likely be necessary to assess the quality of wetlands, where site characteristics are situated within expected ranges of landscape characteristics (Table 3.3).

Table 3.3: Conceptual design for a wetland reclamation criteria table. Wetlands of different types and sizes would be expected to be within the IBI ranges of reference wetlands for a Natural Region. Individual wetlands would be situated in landscapes with given composition and configuration parameters and extents defined by watersheds. Wetland size (ha) ranges are based on the frequency distribution of wetland area. Note that multiple measures would be used for each of the IBI types (i.e. Environmental, Hydrological, Ecological), Topography, and Configuration types (Shape and Aggregation). Diversity, Aggregation, Topography measures are calculated for the entire landscape and thus do not have a row for each wetland type/size

Wetland		**IBI Ranges**			**Topography**	**Composition**			**Configuration**	
Type	Size (ha)	Env.	Hydr.	Eco.		Number of Patches	Percent of landscape	Diversity	Shape	Aggregation
Temporary	< 0.1									
Temporary	0.1 - 1.0									
Temporary	> 1.0									
Seasonal	< 0.1									
Seasonal	0.1 - 1.0									
Seasonal	> 1.0									
Semi-Perm	< 0.1									
Semi-Perm	0.1 - 1.0									
Semi-Perm	> 1.0									
Open Water	< 0.1									
Open Water	0.1 - 1.0									
Open Water	> 1.0									
Alkali	< 0.1									
Alkali	0.1 - 1.0									
Alkali	> 1.0									

Wetland reclamation criteria should include delineation of required ranges of IBI values, landscape composition and configuration, and topography for different types and sizes of wetlands (Table 3.3). Since it is likely that landscape composition, configuration, and topography will vary across large spatial extents, it would be necessary to have a different index table for each Natural Region in Alberta. For large reclamation projects, the design of the landscapes would come before the construction of individual wetlands. Landscape designs could be simulated using composition and configuration parameters based on undisturbed landscapes within the same Natural Region (Brown & Duh, 2004; Duh & Brown, 2007). Characterizing natural configuration and composition allows reclamation planners to target the landscape metric ranges typical of natural landscapes. This should improve the outcomes for wetland reclamation because wetland conditions are affected by the surrounding landscape. Since entire landscapes are being reclaimed, composition and configuration of land covers other than wetlands would have to be considered in the landscape design.

Since composition and configuration of wetland landscapes vary within a Natural Region, a method for determining the parameters for wetland landscape design and simulation requires attention. A simple option would be to base composition and configuration parameters off values calculated from historical imagery of the area pre-disturbance. Most mine sites have high quality pre-disturbance imagery, and using that imagery to calculate baseline metrics would ensure similar composition and configuration of land cover to the original landscape although a slightly different appearance (Deutsch & Cockerham, 1994). However, this method may not be feasible for long term wetland reclamation because returning landscapes to their original state may not be possible due the effect of climate change and legacy disturbance. For instance, the climate of Parkland and Grassland is predicted to expand north over the next century (Schneider, 2013), meaning the climate conditions will likely not be conducive to reclaiming landscapes to their historical composition and configuration. Instead, I argue that that using landscapes in areas that representative of the future environmental conditions of reclaimed landscapes would be more effective, because long term changes in climate are accounted for.

In Chapter 2, a considerable amount of variation in metric values was observed at all disturbance intervals. One challenge of using landscapes metrics to inform reclamation design parameters is selecting the precise metric parameters within the natural range. The simplest option would be to use the median values of the metrics as parameters for landscape simulation.

The drawback to this method is that composition and configuration properties of individually reclaimed landscapes would not replicate the variability of composition and configuration in the Natural Region (Figure 3.2). Coordination between governments and reclamation planners will therefore be necessary to ensure that the variation in configuration and composition of all reclaimed landscapes is comparable to that of natural landscapes.

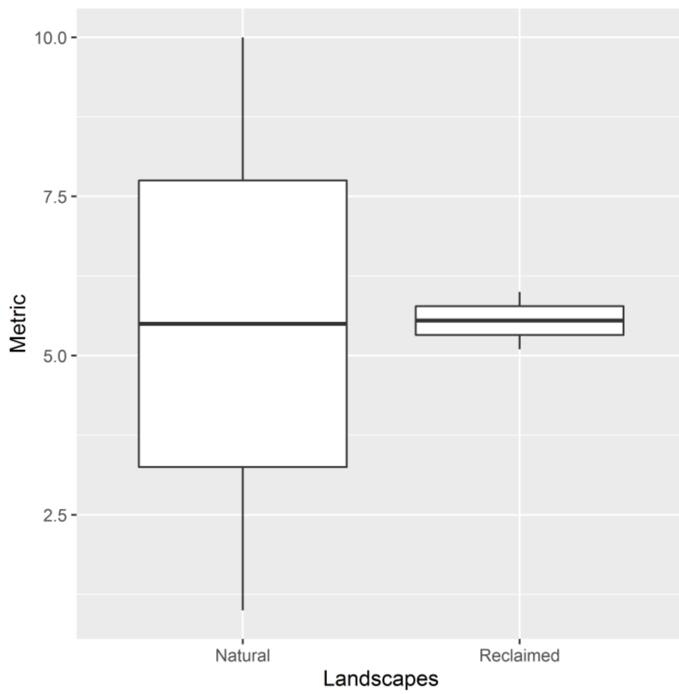

Figure 3.2: Illustration of how using the median (or near-median) values of metric distributions as parameters for individual landscape designs would result in a compressed distributions relative to what occurs naturally.

After a landscape has been designed in terms of composition an configuration, construction would likely begin by shaping the topography since it plays a major role in wetland formation (Lang et al., 2013). Vegetation would then be seeded according to the reclamation and closure plan. Once reclamation has been completed, a regular monitoring program would be necessary to track the success of reclamation. Monitoring could involve site visits to monitor IBI values, and remote sensing to monitor the landscape structure. It is unlikely that reclaimed wetlands and landscapes would initially achieve the same level of functionality, hence the need for long-term monitoring prior to issuance of closure permits. Reclamation could be considered successful if the reclaimed wetlands and landscape achieve and remain within their target ranges over a decades-long time span.

There are practical limitations at all scales of the wetland reclamation process that would impede developer's capability to reclaim a landscape to a state comparable to a natural one. First, site-level wetland reclamation is still experimental. For example, initial fen reclamation efforts in the Alberta Oil Sands are underway but more research is needed to develop a cost-effective manner of monitoring reclaimed fens (Nwaishi et al. 2015). Second, land cover data for quantifying wetland landscape composition and configuration needs to be accurate and consistent across the province to ensure simulated wetland landscapes have functionally and aesthetically similar to natural wetland landscapes. Despite these challenges, wetland reclamation remains imperative because of their immense value to ecosystems and humanity. A holistic approach that incorporates scientific knowledge about individual sites and landscape structure is likely the best way to ensure the long-term sustainability of reclaimed wetlands.

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Appendix A – Correlation Grouping

Table A1: Metrics grouped for Central-All such that within-group Pearson correlations are > |0.9|.

Group	Representative	Other Group Members
1	Patch density of wetlands	
2	Largest patch index of wetlands	Division index of wetlands
3	Area-weighted mean patch area of wetlands	Effective mesh size of wetlands; Area-weighted mean patch area of open water wetlands
4	Area-weighted mean radius of gyration of wetlands	
5	Area-weighted mean shape index of wetlands	Area-weighted mean fractal dimension of wetlands
6	Percent like-adjacencies of wetlands	Area-weighted mean contiguity index of wetlands; Area-weighted mean perimeter-area ratio of wetlands
7	Area-weighted mean related circumscribing circle of wetlands	
8	Area-weighted mean Euclidean nearest neighbour of wetlands	
9	Contagion index of wetlands	
10	Cohesion index of wetlands	
11	Splitting index of wetlands	
12	Patch richness density of wetlands	
13	Simpson's diversity index of wetlands	Shannon's diversity index of wetlands; Shannon's evenness index of wetlands; Modified Simpson's diversity index of wetlands; Simpson's evenness index of wetlands; Modified Simpson's evenness index of wetlands
14	Aggregation index of wetlands	
15	Patch density of alkali wetlands	
16	Area-weighted mean patch area of alkali wetlands	
17	Patch density of seasonal wetlands	
18	Area-weighted mean patch area of seasonal wetlands	

Table A1 (continued)

Group	Representative	Other Group Members
19	Patch density of semi-permanent wetlands	
20	Area-weighted mean patch area of semi-permanent wetlands	
21	Patch density of temporary wetlands	
22	Area-weighted mean patch area of temporary wetlands	
23	Patch density of open water wetlands	
24	Edge density of alkali wetlands	
25	Edge density of seasonal wetlands	
26	Edge density of semi-permanent wetlands	
27	Edge density of temporary wetlands	
28	Edge density of open water wetlands	
29	Edge density of wetlands	
30	Patch richness density of land cover	
31	Simpson's diversity index of land cover	Shannon's diversity index of land cover; Shannon's evenness index of land cover; Modified Simpson's diversity index of land cover; Simpson's evenness index of land cover; Modified Simpson's evenness index of land cover

Table A2: Metrics grouped for Central-Boreal such that within-group Pearson correlations are > |0.9|.

Group	Representative	Other Group Members
1	Patch density of wetlands	
2	Largest patch index of wetlands	Division index of wetlands
3	Area-weighted mean patch area of wetlands	Effective mesh size of wetlands; Area-weighted mean patch area of open water wetlands
4	Area-weighted mean radius of gyration of wetlands	
5	Area-weighted mean shape index of wetlands	Area-weighted mean fractal dimension of wetlands
6	Percent-like adjacencies of wetlands	Area-weighted mean contiguity index of wetlands; Area-weighted mean perimeter-area ratio of wetlands
7	Area-weighted mean related circumscribing circle of wetlands	
8	Area-weighted mean Euclidean nearest neighbour of wetlands	
9	Contagion index of wetlands	
10	Cohesion index of wetlands	
11	Splitting index of wetlands	
12	Patch richness density of wetlands	
13	Simpson's diversity index of wetlands	Shannon's diversity index of wetlands; Shannon's evenness index of wetlands; Modified Simpson's diversity index of wetlands; Simpson's evenness index of wetlands; Modified Simpson's evenness index of wetlands
14	Aggregation index of wetlands	
15	Patch density of alkali wetlands	
16	Area-weighted mean patch area of alkali wetlands	
17	Patch density of seasonal wetlands	
18	Area-weighted mean patch area of seasonal wetlands	
19	Patch density of semi-permanent wetlands	
20	Area-weighted mean patch area of semi-permanent wetlands	

Table A2 (continued)

Group	Representative	Other Group Members
21	Patch density of temporary wetlands	
22	Patch density of open water wetlands	
23	Edge density of alkali wetlands	
24	Edge density of seasonal wetlands	
25	Edge density of semi-permanent wetlands	
26	Edge density of temporary wetlands	
27	Edge density of open water wetlands	
28	Edge density of wetlands	
29	Patch richness density of land cover	
30	Simpson's diversity index of land cover	Shannon's diversity index of land cover; Shannon's evenness index of land cover; Modified Simpson's diversity index of land cover; Simpson's evenness index of land cover; Modified Simpson's evenness index of land cover

Table A3: Metrics grouped for Central-Parkland such that within-group Pearson correlations are > |0.9|.

Group	Representative	Other Group Members
1	Patch density of wetlands	
2	Largest patch index of wetlands	Division index of wetlands
3	Area-weighted mean patch area of wetlands	
4	Area-weighted mean radius of gyration of wetlands	
5	Area-weighted mean shape index of wetlands	Area-weighted mean fractal dimension of wetlands
6	Percent like-adjacencies of wetlands	Area-weighted mean contiguity index of wetlands; Area-weighted mean perimeter-area ratio of wetlands
7	Area-weighted mean related circumscribing circle of wetlands	
8	Area-weighted mean Euclidean nearest neighbour of wetlands	
9	Contagion index of wetlands	
10	Cohesion index of wetlands	
11	Effective mesh size of wetlands	
12	Splitting index of wetlands	
13	Patch richness density of wetlands	
14	Simpson’s diversity index of wetlands	Shannon’s diversity index of wetlands; Shannon’s evenness index of wetlands; Modified Simpson’s diversity index of wetlands; Simpson’s evenness index of wetlands; Modified Simpson’s evenness index of wetlands
15	Aggregation index of wetlands	
16	Patch density of alkali wetlands	
17	Area-weighted mean patch area of alkali wetlands	
18	Patch density of seasonal wetlands	
19	Area-weighted mean patch area of seasonal wetlands	
20	Patch density of semi-permanent wetlands	
21	Area-weighted mean patch area of semi-permanent wetlands	

Table A3 (continued)

Group	Representative	Other Group Members
22	Patch density of temporary wetlands	
23	Area-weighted mean patch area of temporary wetlands	
24	Patch density of open water wetlands	
25	Area-weighted mean patch area of open water wetlands	
26	Edge density of alkali wetlands	
27	Edge density of seasonal wetlands	
28	Edge density of semi-permanent wetlands	
29	Edge density of temporary wetlands	
30	Edge density of open water wetlands	
31	Edge density of wetlands	
32	Patch richness density of land cover	
33	Simpson's diversity index of land cover	Shannon's diversity index of land cover; Shannon's evenness index of land cover; Modified Simpson's diversity index of land cover; Simpson's evenness index of land cover; Modified Simpson's evenness index of land cover

Table A4: Metrics grouped for Southern-All such that within-group Pearson correlations are > |0.9|.

Group	Representative	Other Group Members
1	Patch density of wetlands	
2	Largest patch index of wetlands	
3	Area-weighted mean patch area of wetlands	Effective mesh size of wetlands
4	Area-weighted mean radius of gyration of wetlands	
5	Area-weighted mean shape index of wetlands	Area-weighted mean fractal dimension of wetlands
6	Percent like-adjacencies of wetlands	Area-weighted mean contiguity index of wetlands; Area-weighted mean perimeter-area ratio of wetlands
7	Area-weighted mean related circumscribing circle of wetlands	
8	Area-weighted mean Euclidean nearest neighbour of wetlands	
9	Contagion index of wetlands	
10	Cohesion index of wetlands	
11	Division index of wetlands	
12	Splitting index of wetlands	
13	Patch richness density of wetlands	
14	Simpson's diversity index of wetlands	Shannon's diversity index of wetlands; Shannon's evenness index of wetlands; Modified Simpson's diversity index of wetlands; Simpson's evenness index of wetlands; Modified Simpson's evenness index of wetlands
15	Aggregation index of wetlands	
16	Patch density of alkali wetlands	
17	Area-weighted mean patch area of alkali wetlands	
18	Patch density of seasonal wetlands	
19	Area-weighted mean patch area of seasonal wetlands	
20	Patch density of semi-permanent wetlands	Edge density of semi-permanent wetlands
21	Area-weighted mean patch area of semi-permanent wetlands	

Table A4 (continued)

Group	Representative	Other Group Members
22	Patch density of temporary wetlands	
23	Area-weighted mean patch area of temporary wetlands	
24	Patch density of open water wetlands	
25	Area-weighted mean patch area of open water wetlands	
26	Edge density of alkali wetlands	
27	Edge density of seasonal wetlands	
28	Edge density of temporary wetlands	
29	Edge density of open water wetlands	
30	Edge density of wetlands	
31	Patch richness density of land cover	
32	Simpson's diversity index of land cover	Shannon's diversity index of land cover; Shannon's evenness index of land cover; Modified Simpson's diversity index of land cover; Simpson's evenness index of land cover; Modified Simpson's evenness index of land cover

Table A5: Metrics grouped for Southern-Parkland such that within-group Pearson correlations are > |0.9|.

Group	Representative	Other Group Members
1	Patch density of wetlands	
2	Largest patch index of wetlands	Division index of wetlands
3	Area-weighted mean patch area of wetlands	Effective mesh size of wetlands
4	Area-weighted mean radius of gyration of wetlands	
5	Area-weighted mean shape index of wetlands	Area-weighted mean fractal dimension of wetlands
6	Percent like-adjacencies of wetlands	Contiguity index of wetlands; Area-weighted mean perimeter-area ratio of wetlands
7	Area-weighted mean related circumscribing circle of wetlands	
8	Area-weighted mean Euclidean nearest neighbour of wetlands	
9	Contagion index of wetlands	
10	Cohesion index of wetlands	
11	Splitting index of wetlands	
12	Patch richness density of wetlands	
13	Simpson's diversity index of wetlands	Shannon's diversity index of wetlands; Shannon's evenness index of wetlands Modified Simpson's diversity index of wetlands; Simpson's evenness index of wetlands; Modified Simpson's evenness index of wetlands
14	Aggregation index of wetlands	
15	Patch density of alkali wetlands	Edge density of alkali wetlands
16	Area-weighted mean patch area of alkali wetlands	
17	Patch density of seasonal wetlands	
18	Area-weighted mean patch area of seasonal wetlands	
19	Patch density of semi-permanent wetlands	Edge density of semi-permanent wetlands
20	Area-weighted mean patch area of semi-permanent wetlands	
21	Patch density of temporary wetlands	

Table A5 (continued)

Group	Representative	Other Group Members
22	Area-weighted mean patch area of temporary wetlands	
23	Patch density of open water wetlands	
24	Area-weighted mean patch area of open water wetlands	
25	Edge density of seasonal wetlands	
26	Edge density of temporary wetlands	
27	Edge density of open water wetlands	
28	Edge density of wetlands	
29	Patch richness density of land cover	
30	Simpson’s diversity index of land cover	Shannon’s diversity index of land cover; Shannon’s evenness index of land cover; Modified Simpson’s diversity index of land cover; Simpson’s evenness index of land cover; Modified Simpson’s evenness index of land cover

Table A6: Metrics grouped for Southern-Grassland such that within-group Pearson correlations are > |0.9|.

Group	Representative	Other Group Members
1	Patch density of wetlands	
2	Largest patch index of wetlands	Division index of wetlands
3	Area-weighted mean patch area of wetlands	Effective mesh size of wetlands
4	Area-weighted mean radius of gyration of wetlands	
5	Area-weighted mean shape index of wetlands	Area-weighted mean fractal dimension of wetlands
6	Percent like-adjacencies of wetlands	Area-weighted mean contiguity index of wetlands; Area-weighted mean perimeter-area ratio of wetlands
7	Area-weighted mean related circumscribing circle of wetlands	
8	Area-weighted mean Euclidean nearest neighbour of wetlands	
9	Contagion index of wetlands	
10	Cohesion index of wetlands	
11	Splitting index of wetlands	
12	Patch richness density of wetlands	
13	Simpson's diversity index of wetlands	Shannon's diversity index of wetlands; Shannon's evenness index of wetlands; Modified Simpson's diversity index of wetlands; Simpson's evenness index of wetlands; Modified Simpson's evenness index of wetlands
14	Aggregation index of wetlands	
15	Patch density of alkali wetlands	
16	Area-weighted mean patch area of alkali wetlands	
17	Patch density of seasonal wetlands	
18	Area-weighted mean patch area of seasonal wetlands	
19	Patch density of semi-permanent wetlands	Edge density of semi-permanent wetlands
20	Area-weighted mean patch area of semi-permanent wetlands	
21	Patch density of temporary wetlands	

Table A6 (continued)

Group	Representative	Other Group Members
22	Area-weighted mean patch area of temporary wetlands	
23	Patch density of open water wetlands	
24	Area-weighted mean patch area of open water wetlands	
25	Edge density of alkali wetlands	
26	Edge density of seasonal wetlands	
27	Edge density of temporary wetlands	
28	Edge density of open water wetlands	
29	Edge density of wetlands	
30	Patch richness density of land cover	
31	Simpson's diversity index of land cover	Shannon's diversity index of land cover; Shannon's evenness index of land cover; Modified Simpson's diversity index of land cover; Simpson's evenness index of land cover; Modified Simpson's evenness index of land cover

Appendix B – PCA Factor Loading Tables

Table B1: Factor loadings for Central-All. Bolded values are representative metrics

Metric	Comp.1	Comp.2	Comp.3
SHAPE_WET	**-0.653**	0.074	-0.039
CIRCLE_WET	-0.58	0.244	-0.27
ENN_WET	0.106	0.515	0.361
CONTAG_WET	-0.148	-0.262	0.528
COHES_WET	-0.412	-0.446	0.101
SIDI_WET	-0.177	**0.574**	0.392
SIDI_LAND	-0.053	-0.268	**0.594**
Eigenvalue	1.791	1.302	1.108
Proportion of variance (%)	25.579	18.599	15.832
Cum. prop. of variance (%)	25.579	44.177	60.01

Table B2: Factor loadings for Central-Boreal. Bolded values are representative metrics. Note: PD_OW and SIDI_LAND are representatives for component 3.

Metric	Comp.1	Comp.2	Comp.3	Comp.4
SHAPE_WET	**-0.643**	-0.011	-0.12	0.027
CIRCLE_WET	-0.585	-0.232	-0.206	0.186
ENN_WET	0.1	-0.414	-0.248	**-0.68**
CONTAG_WET	-0.122	0.453	0.215	-0.513
COHES_WET	-0.402	0.433	0.101	-0.145
SIDI_WET	-0.223	**-0.584**	0.278	-0.209
PD_OW	0.016	-0.202	**0.611**	0.337
SIDI_LAND	-0.088	-0.012	**0.612**	-0.244
Eigenvalue	1.778	1.376	1.168	1.027
Proportion of variance (%)	22.229	17.201	14.598	12.839
Cum. prop. of variance (%)	22.229	39.43	54.028	66.868

Table B3: Factor loadings for Central-Parkland. Bolded values are representative metrics

Metric	Comp.1	Comp.2	Comp.3
SHAPE_WET	-0.325	-0.586	-0.017
CIRCLE_WET	-0.187	**-0.611**	0.232
ENN_WET	0.138	0.005	-0.48
CONTAG_WET	-0.102	-0.117	-0.532
COHES_WET	**-0.674**	0.041	0.044
SIDI_WET	0.114	-0.282	**-0.604**
AI_WET	-0.544	0.356	-0.253
SIDI_LAND	-0.258	0.248	-0.043
Eigenvalue	1.963	1.777	1.247
Proportion of variance (%)	24.535	22.209	15.59
Cum. prop. of variance (%)	24.535	46.744	62.334

Table B4: Factor loadings for Southern-All. Bolded values are representative metrics

Metric	Comp.1	Comp.2	Comp.3	Comp.4
CIRCLE_WET	-0.05	0.158	**0.76**	-0.376
ENN_WET	-0.275	-0.155	-0.309	-0.432
CONTAG_WET	-0.166	**-0.611**	0.253	0.129
COHESION_WET	**0.637**	-0.046	0.068	-0.108
SPLIT_WET	-0.27	0.328	-0.099	**0.614**
SIDI_WET	-0.309	-0.583	-0.015	0.035
AI_WET	0.551	-0.323	-0.246	0.117
SIDI_LAND	0.129	-0.154	0.433	0.502
Eigenvalue	2.111	1.487	1.192	1.048
Proportion of variance (%)	26.382	18.586	14.897	13.1
Cum. prop. of variance (%)	26.382	44.968	59.865	72.965

Table B5: Factor loadings for Southern-Parkland. Bolded values are representative metrics

Metric	Comp.1	Comp.2	Comp.3
CIRCLE_WET	0.22	0.361	0.072
ENN_WET	-0.346	-0.006	-0.417
CONTAG_WET	-0.317	-0.478	0.18
COHES_WET	**0.605**	-0.24	-0.192
SIDI_WET	-0.439	-0.403	-0.128
AI_WET	0.409	**-0.604**	-0.256
SIDI_LAND	0.076	-0.237	**0.818**
Eigenvalue	2.097	1.419	1.06
Proportion of variance (%)	29.96	20.274	15.148
Cum. prop. of variance (%)	29.96	50.235	65.382

Table B6: Factor loadings for Southern-Grassland. Bolded values are representative metrics

Metric	Comp.1	Comp.2	Comp.3	Comp.4
CIRCLE_WET	-0.1	0.029	**0.771**	0.374
ENN_WET	-0.256	-0.128	-0.359	0.439
CONTAG_WET	-0.143	**-0.641**	0.16	-0.149
COHESION_WET	**0.634**	-0.035	0.094	0.103
SPLIT_WET	-0.29	0.34	-0.052	**-0.621**
SIDI_WET	-0.285	-0.596	-0.1	-0.075
AI_WET	0.566	-0.26	-0.261	-0.134
SIDI_LAND	0.131	-0.179	0.402	-0.474
Eigenvalue	2.136	1.492	1.214	1.038
Proportion of variance (%)	26.698	18.649	15.178	12.976
Cum. prop. of variance (%)	26.698	45.348	60.526	73.502

Appendix C – Descriptions of Representative Metrics

Table C1: Formulas, units, and ranges for the selected representative metrics

Type	Metric	Abbrv.	Formula	Units	Range
Shape	Area-weighted mean shape index	SHAPE	$\sum_{i=1}^{m}\sum_{j=1}^{n}\left[\frac{p_{ij}}{\min p_{ij}}\left(\frac{a_{ij}}{\sum_{i=1}^{m}\sum_{j=1}^{n}a_{ij}}\right)\right]$	none	SHAPE ≥ 1
	Area-weighted mean related circumscribing circle	CIRCLE	$\sum_{i=1}^{m}\sum_{j=1}^{n}\left[\frac{a_{ij}}{a_{ij}^{s}}\left(\frac{a_{ij}}{\sum_{i=1}^{m}\sum_{j=1}^{n}a_{ij}}\right)\right]$	none	0 ≤ CIRCLE ≤ 1
Aggregation	Aggregation index	AI	$\left[\sum_{i=1}^{m}\left(\frac{g_{ii}}{\max g_{ii}}\right)P_i\right]100$	%	0 ≤ AI ≤ 100
	Contagion index	CONTAG	$\left[1+\frac{\sum_{i=1}^{m}\sum_{j=1}^{m}\left[P_i\left(\frac{g_{ij}}{\sum_{j=1}^{m}g_{ij}}\right)\right]\cdot\left[\ln(P_i)\left(\frac{g_{ij}}{\sum_{j=1}^{m}g_{ij}}\right)\right]}{2\ln(m)}\right]\cdot 100$	%	0 < CONTAG ≤ 100
	Patch cohesion index	COHES	$\left(1-\frac{\sum_{i=1}^{m}\sum_{j=1}^{n}p_{ij}}{\sum_{i=1}^{m}\sum_{j=1}^{n}p_{ij}\sqrt{a_{ij}}}\right)\left(1-\frac{1}{\sqrt{Z}}\right)^{-1}\cdot 100$	%	0 < COHES < 100
	Area-weighted mean Euclidean nearest neighbour	ENN	$\sum_{i=1}^{m}\sum_{j=1}^{n}\left[h_{ij}\left(\frac{a_{ij}}{\sum_{i=1}^{m}\sum_{j=1}^{n}a_{ij}}\right)\right]$	m	ENN > 0
	Patch density	PD	$\left(\frac{n_i}{A}\right)(10{,}000)(100)$	Number/ 100 ha	PD ≥ 0

Table C1 (continued)

Type	Metric	Abbrv.	Formulas	Units	Ranges
Aggr.	Splitting index	SPLIT	$\frac{A^2}{\sum_{i=1}^{m}\sum_{j=1}^{m} {a_{ij}}^2}$	none	1 ≤ SPLIT ≤ (Number of cells in landscape)^2
Diversity	Simpson's diversity index	SIDI	$1 - \sum_{i=1}^{m} {P_i}^2$	none	0 ≤ SIDI < 1

$\boldsymbol{a_{ij}}$ = area of patch *ij*
$\boldsymbol{a_{ij}^s}$ = area of small circle circumscribing patch a_{ij}
$\boldsymbol{A}$ = total area of landscape
$\boldsymbol{e_{ij}}$ = total length of edge involving class *i*
$\boldsymbol{p_{ij}}$ = perimeter of patch *ij*
$\boldsymbol{n_i}$ = number of patches for class *i*
$\boldsymbol{h_{ij}}$ = distance from patch *ij* to nearest patch of the same class (edge to edge distance)
$\boldsymbol{g_{ii}}$ = number of like-adjacencies for pixels of class *i*
$\boldsymbol{g_{ij}}$ = number of adjacencies between pixels of classes *i* and *j*
$\boldsymbol{P_i}$ = proportion of landscape occupied by class *i*
$\boldsymbol{Z}$ = number of cells in the landscape
$\boldsymbol{m}$ = number of classes

Appendix D – Representatives Using Relaxed Criteria

Table D1: Comparison of representative metrics for Natural Region subsets in the Central inventory using relaxed criteria. Instead of taking only the metric with the highest absolute loading on each component, the relaxed method retained all metrics with a loading > |0.5|.

Metric	Central Boreal	Central Parkland	Common
SHAPE_WET	x	x	x
CIRCLE_WET	x	x	x
ENN_WET	x		
CONTAG_WET	x	x	x
COHES_WET		x	
SIDI_WET	x	x	x
PD_OW	x		
SIDI_LAND	x		
AI_WET		x	

Table D2: Comparison of representative metrics for Natural Region subsets in the Central inventory using relaxed criteria. Instead of taking only the metric with the highest absolute loading on each component, the relaxed method retained all metrics with a loading > |0.5|.

Metric	Southern Parkland	Southern Grassland	Common
COHES_WET	x	x	x
AI_WET	x	x	x
SIDI_LAND	x		
CONTAG_WET		x	
COHES_WET		x	
SPLIT_WET		x	
SIDI_WET		x	

Appendix E – Land Cover Summary by Natural Region

Table E1: Land cover summarized by Natural Region using the 2009 AAFC data. All values are given in percentages.

Land Cover	Boreal	Parkland	Grassland
Water	3.78	1.87	1.57
Exposed	0.69	0.55	1.19
Developed	0.35	3.16	1.06
Shrub	7.28	5.08	1.48
Wetland	15.89	2.17	2.21
Grass	0.21	7.48	41.96
Agriculture	18.65	75.05	50.12
Forest	53.15	4.64	0.40

Appendix F – Representatives Using Residual Metrics

Table F1: Representative metrics for the six spatial subsets. Italicized metrics are residual metrics.

Principal Component	Central-All	Central-Boreal	Central-Parkland	Southern-All	Southern-Parkland	Southern-Grassland
1	*ED_WET* *Area/Edge*	*ED_WET* *Area/Edge*	*PD_WET* *Aggregation*	*PD_WET* *Aggregation*	*GYRATE_WET* *Area/Edge*	*PD_WET* *Aggregation*
2	SHAPE_WET Shape	SHAPE_WET Shape	AI_WET Aggregation	*ED_ALKA* *Area/Edge*	CIRCLE_WET Shape	*SHAPE_WET* *Shape*
3	COHES_WET Aggregation	COHES_WET Aggregation	COHES_WET Aggregation	*SHAPE_WET* *Shape*	*SHAPE_WET* *Shape*	*AREA_OW* *Area/Edge*
4	*ED_TEMP* *Area/Edge*	*ED_ALKA* *Area/Edge*	*AREA_TEMP* *Area/Edge*	*ED_ALKA* *Area/Edge*	*PD_TEMP* *Aggregation*	*ED_ALKA* *Area/Edge*
5	ED_ALKA Area/Edge	PD_OW Aggregation	*ED_OW* *Area/Edge*	*PD_SEMIP* *Aggregation*	SIDI_WET Diversity	COHES_WET Aggregation
6	*PD_SEAS* *Aggregation*	*ED_ALKA* *Area/Edge*	*ED_SEMIP* *Area/Edge*	*AREA_TEMP* *Area/Edge*	*PD_SEMIP* *Aggregation*	CIRCLE_WET Shape
7	*LPI_WET* *Area/Edge*	*ED_OW* *Area/Edge*	*ED_ALKA* *Area/Edge*	SIDI_WET Diversity	SIDI_LAND Diversity	SIDI_WET Diversity
8	*ED_OW* *Area/Edge*	CONTAG_WET Aggregation	CONTAG_WET Aggregation	*PD_SEMIP* *Aggregation*	*SPLIT_WET* *Aggregation*	*PD_SEMIP* *Aggregation*
9	SIDI_LAND Diversity	ENN_WET Aggregation	*ED_TEMP* *Area/Edge*	*DIVIS_WET* *Aggregation*	*AREA_ALKA* *Area/Edge*	SIDI_LAND Diversity
10	ENN_WET Aggregation		CONTAG_WET Aggregation	SPLIT_WET Aggregation	*PD_ALKA* *Aggregation*	ED_OW Area/Edge

Appendix G – Comparing COHES_WET Between Parkland Subsets

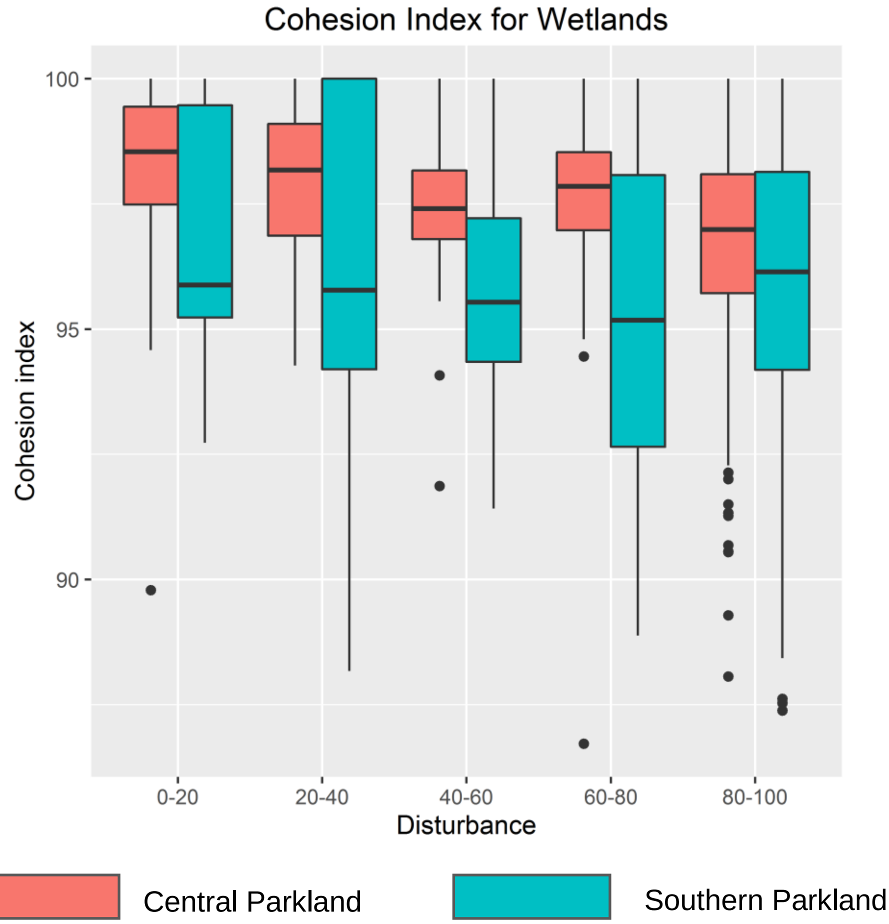

Figure F1: Comparison of COHES_WET values between the Parkland subsets of in the Central and Southern wetland inventories for each disturbance level.

Table F1: Significance levels (p-values) of the Kruskal-Wallis comparisons of metric distributions between Natural Regions (Boreal and Parkland) at corresponding disturbance levels in the Central inventory.

	Disturbance (%)				
Metric	0-20	20-40	40-60	60-80	80-100
COHES_WET	0.055	0.136	0.011*	<0.001***	0.005*