An Exploratory Methodology for Quantifying Land-cover Patterns Along Permanent Open-water and Disturbance Levels for Large-Scale Wetland Reclamation

by

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Author's Declaration

I hereby declare that I am the sole author of this thesis. This is a true copy of the thesis, including any required final revisions, as accepted by my examiners. I understand that my thesis may be made electronically available to the public.

Statement of Contributions

While it is true that I am the sole author of this thesis, conducted all the analysis and did all the writing, I worked closely with my supervisor Dr. Derek Robinson, who met with me regularly to brainstorm ideas and edited my work several times. His guidance was instrumental to this project and to acknowledge that, I use the terms “we/our” throughout the document. Further, this research is part of a larger interdisciplinary effort involving multiple departments at the University of Waterloo and other universities. Therefore, I feel it is a bit outmoded to have to declare myself the sole author when so many people contributed to my ability to complete this project.

Abstract

Wetlands are multi-functional systems that provide a disproportionate number of ecosystem services given the spatial extent they occupy both nationally and globally. The ecological functioning of these wetlands is dependent on the structure of the landscape, which poses unique challenges when reclaiming wetlands in areas where resource extraction is occurring. Resource extraction mega-projects require that entire landscapes be reclaimed and often involve timelines that necessitate the consideration of climate projections to create self-sustaining, naturally appearing wetlands, that integrate with the broader landscape and meet policy objectives. A parsimonious set of landscape metrics was applied to 13,676–1km^2 random sample landscapes to quantify the variation in the composition and configuration of land-cover. Landscape metric values were compared across levels of the proportion of permanent open-water wetland (up to 20%), anthropogenic disturbance and across natural regions (i.e., Grassland, Parkland and Boreal). Results demonstrate statistical differences between landscapes comprising 0-80% and 80-99.9% disturbance in the Boreal and Parkland regions and statistically significant differences among the 0-20% disturbed landscapes in the Grassland region. While differences in landscape pattern were present among the disturbance levels between 0-80% in the Boreal and Parkland region, these were less systematic. Further, the majority (>85%) of permanent open-water wetlands in our samples were found to have less than or equal to eight percent (0.08km^2) of their total area classified as permanent open-water wetland, which is a smaller proportion than what is typically found in closure plans. This exploratory method highlights that permanent open-water alone is inadequate to capture changes resulting from anthropogenic disturbances in wetland-rich landscapes and that regulators should to enforce the creation of multiple wetland types and consider climate change in closure plans. We discuss our results, issues, the novelty of applying such methods to landscape-level reclamation and make suggestions for further work.

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Dedication

The first dedication is to my future self. You're welcome.

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With love,
Jenny

p.s. It's been a while since you have read anything I wrote, and I think you'll be pleased to find a significant improvement in my grammar and sentence structure ;)

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Chapter 1—Introduction

1.1 Defining Wetlands

Wetlands occur at the interface of terrestrial and aquatic systems, making them inherently variable and complex (Marton et al. 2015). This complexity can create a multitude of wetland types that range from permanently flooded coastal mangroves to smaller inland depressions that are saturated for a few months a year. Further, wetland hydrology is influenced by local climate and watershed characteristics, which presents a challenge in defining wetland types and impedes the creation of a universally accepted wetland classification system (Keddy, 2010). However, wetlands can be broadly defined by their relationship to the water table, whereby wetlands are areas that are occasionally or continually inundated by water and contain vegetation adapted for anaerobic environments and hydric soils (Winter, 2000). Although this broad definition is commonly accepted, each country has its own legal definition and regional classification systems are often dictated by geographic conditions. National and regional classifications are typically oriented towards better conservation, management and assessment based on the priorities of the level they are applied (Finlayson & van der Valk, 1996). While these localized classification systems provide a crucial first step for inventorying and managing wetlands, resolving the differences among them would be required for any international system to be fully implemented and accepted (Finlayson & van der Valk, 1996).

The first documented attempt to create an internationally accepted wetland classification system came from the RAMSAR Convention on Wetlands of International Importance (1971). The RAMSAR classification system recognizes 30 wetlands types and is divided into three major categories: marine/coastal, inland and human-made. These are further sub-divided by water type: fresh, saline, brackish and alkaline (Matthews, 1993). Wetlands are defined in this system as: "areas of marsh, fen, peatland or water, whether natural or artificial, permanent or temporary, with water that is static or flowing, fresh, brackish or salt, including areas of marine water, the depth of which at low tide does not exceed six meters" (Matthews, 1993, pg. 38). Though this definition has been agreed to by 75 of the Contracting Parties of the Convention, it does not imply total acceptance and is often a source of local controversy when attempting to determine if a habitat is or is not a wetland (Hughes, 1995; Finlayson & van der Valk, 1996). For instance, the inclusion of coral reefs and the depth criteria of six meters is often debated by those working in marine wetland environments as coral reefs often extend beyond this cut-off, which causes delineation issues (Lu, 1995). The debate over delineation or average size in wetland classifications also plagues those classification systems used at the national or regional level, highlighting the challenge for achieving consistency and satisfying the needs of stakeholders when developing wetland classification systems at any level (Cowardin & Golet, 1995).

In Canada, our wetland classification system was influenced by the RAMSAR system and those already in existence throughout Europe and Scandinavia (e.g. Bellamy, 1968); however, the Canadian system attempted to evolve beyond the traditional approaches in wetland classification and develop a system within a multi-disciplinary framework that reduced the need for users to have a strong background in fields related to wetland research (e.g., hydrology or vegetation) (Wells & Zoltai, 1985).

1.2 Canadian wetland classification system

The earliest attempt at a national wetland classification system in Canada occurred in 1973, as part of an Organic Terrain Classification system by the National Committee on Forest Lands (NWWG, 1997). This initial attempt prompted researchers to classify and inventory wetlands across Canada and led to the proposal of a four-tiered, ecologically-based wetland classification system that consisted of four levels: wetland classes (e.g., bog), wetland form (e.g., domed bogs), vegetation, and specialization (e.g., engineering needs, forestry) (Wells & Zoltai, 1985). The basis of this preliminary work was developed into a more extensive national system and gave way to several refined regional wetland classification systems: Ontario (Jeglum et al. 1974), the Prairies (Millar, 1976), British Columbia (Runka & Lewis, 1981), Quebec (Couillard & Grondin, 1986), and northern Ontario (Harris et al. 1996). The first provisional edition of the Canadian Wetland Classification System (CWCS) was released in 1987 (NWWG, 1997) and differed in its basic philosophy from other classification systems in existence at the time by using wetland functions as the basis (i.e., the interrelationships of abiotic and biotic components of wetland ecosystems) (Zolati & Vitt, 1995). From this initial publication, the classification was refined in a second edition that was designed to transcend regional differences and provide flexibility. The latest edition has three basic levels of classification: class, form, and type, which is further divided into two categories: organic or mineral, that are further sub-divided into five classes by their biotic properties, hydrology and nutrient supply (e.g., fens, bogs, swamps marshes and open-water) (Figure 1) (NWWG, 1997).

Organic wetlands (i.e., bogs and fens) are characterized by the accumulation of peat (i.e., decaying organic mater) that is larger than 40 cm and permanently saturated. As a result, organic wetlands are often referred to as "peatlands", which can be sub-divided by dominant water source and acidity level. Minerotrophic wetlands (i.e., fens) are fed mainly by groundwater, while ombrotrophic wetlands (i.e., bogs) are sustained mainly from precipitation, are nutrient poor, are more acidic than fens, and dominated by *Sphagnum* moss (NWWG, 1997). In contrast to organic wetlands, mineral wetlands (i.e., marshes, swamps and open-water wetlands) tend to be characterized by their vegetation and limited peat accumulation. Marshes have herbaceous emergent vegetation and are flooded periodically, whereas swamps have prolonged periods of saturation and vegetation that is dominated by trees. Shallow open-water wetlands have standing water all year round with a depth of approximately two meters in midsummer, whereas open-water wetlands are often described as larger bodies of water that represent the transition between marshes and lakes. While similar in their definitions, shallow open-water wetlands can have emergent and submerged aquatic vegetation, but open-water wetlands are limited to submerged aquatic vegetation (NWWG, 1997; Wray & Bayley, 2006). It should be noted that bogs and fens are the dominant peatland classes, though some swamps and marshes in the Boreal region can also accumulate peat (Bayley & Mewhort, 2004). The CWCS is utilized nationally and is similar to classification systems used in the United States (U.S.). Like Canada, the U.S. uses several classification systems, some of which can be applied on either side of the border.

1.3 A brief review of American wetland classification systems

Wetlands in the U.S. are typically classified using the Cowardin System (Cowardin et al. 1979). The Cowardin (1979) system is one of the most comprehensive wetland classifications in the U.S. and divides wetlands into systems, sub-systems, classes, and sub-classes, across a series of hydrologic regimes, chemistry and soil modifiers (Finlayson & van der Valk, 1995). The five major wetland types within this system are: marine, tidal, lacustrine, palustrine, and riverine (EPA, 2018). These are further divided based on frequency of inundation and then classed based on their position in the landscape, hydrologic regime, and vegetation cover (Table 1). The Cowardin system is used by the National Wetland Inventory and U.S. Fish and Wildlife Service. In contrast, the U.S. Army Corps of Engineers uses a classification system based on the underlying hydrology and unifying landforms, which is designed to assess the physical, chemical and biological functions of wetlands (Brinson, 1993). Like the Cowardian (1979) system, Brinson (1993) classifies wetlands into five wetland types (riverine, slope, depressional, flat, and fringe) but does so by their hydrogeomorphic (HGM) units (i.e., location, hydrodynamics and the dominate water source). A benefit of this system is that it can be used for both coastal and inland wetlands, regardless of vegetation types and climate (Brinson, 1993).

Another classification system coming out of the U.S. that is referred to as the Stewart and Kantrud system (1971) was created specifically for the formerly glaciated prairie pothole region and is often applied in southern Canada due to the similar climate and history (Stewart & Kantrud, 1971; Wray & Bayley, 2006). The prairie pothole region is characterized by millions of depressional wetlands that span the Great Plains and Central Lowlands of the U.S. and extends into the southern Alberta, Saskatchewan, and southwest Manitoba regions of Canada. In the Stewart & Kantrud (1971) system, wetlands are grouped by vegetation zones: wetland-low-prairie, wet-meadow, shallow-marsh, deep-marsh, permanent-open-water, intermittent-alkali zone, and fen (alkaline bog) zones (Stewart & Kantrud, 1996) (Table 1).

Table 1: Comparison of the Stewart & Kantrud (1971) and the Cowardin (1979) wetland classification systems.

Stewart & Kantrud	**Cowardin**
Low-prairie	Rarely flooded
Wet meadow (if dried out)	Temporarily flooded
Wet meadow	Seasonally flooded
Shallow Marsh	Semi-permanently flooded
Deep marsh	Intermittently exposed
Intermittent-alkaline	Intermittently flooded
Permanent-open-water	Permanently flooded
Fen (alkaline bog)	Saturated

While a universal wetland classification system would be beneficial, the geographically specific effects of climate change on wetlands will likely further the reliance on more localized classification systems. For example, in Canada, the Grassland region is expected to see the largest increases in annual temperature due to climate change and a northward shift in the natural regions is expected (Erwin, 2009; Schneider et al. 2015). Therefore, it may be more useful for modelling and classifying wetlands to focus on the climate envelope in which wetlands occur and their hydroperiod (i.e. intermittent, temporary, seasonal, semi-permanent and permanent) to manage them more efficiently in a changing climate.

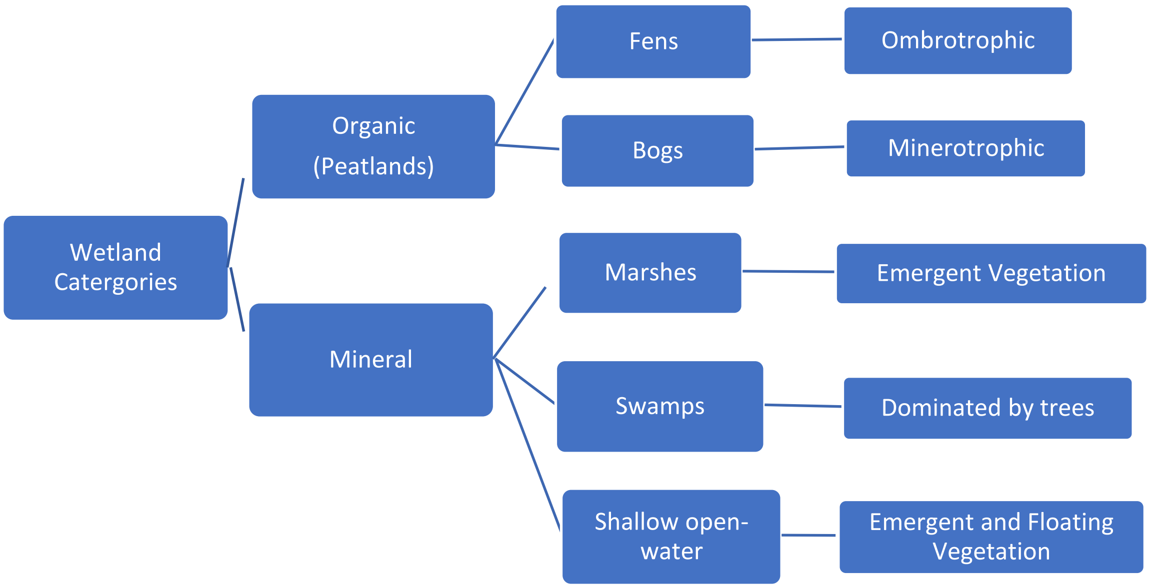


Figure 1: General description of wetland categories adapted from the Canadian Wetland Classification System (National Wetlands Working Group, 1997)

2. The Value of Wetlands

Historically, wetlands were believed to be an impediment to productive agriculture activities, which led to filling or draining them to "improve" the land (Keddy, 2010). Wetlands in North America never held a position of economic importance like they did in some European and Scandinavian countries, where peat had been utilized for decades as a source of energy for heating purposes (Wells & Zoltai, 1985). However, wetlands are now recognized as one of the most valuable and productive ecosystems on earth; comparable to coral reefs and rain forests (Mitsch & Gosselink, 2000). Despite only occupying approximately 6-9% of world's land surface (Matthews & Fung, 1987; Zedler & Kercher, 2005; Erwin, 2009), it has been estimated that wetlands provide an array of ecosystem services that accounts for 25% of global productivity (Zedler & Kercher, 2005; Ducks Unlimited, 2006).

Wetlands are often referred to as the "earth's kidneys" due to the crucial hydrological, biological and chemical functions they can perform (Table 2) (Mitsch & Gosselink, 1993), and the significant value of these functions to society (Keddy, 2010). For example, wetlands may remove pollutants and purify water (Tournebize et al. 2017), filter sediment (Kuenzler, 1989), recharge ground water (van der Kamp & Hayashi, 1998), mitigate floods (Ming et al. 2007) and droughts (Hood & Bayley, 2008), act as long-term carbon sinks (Burkett & Kusler, 2000; Kayranli et al. 2010), support biodiversity, and provide habitat to rare and at-risk species (Keddy, 2010). In addition to these potential ecological and hydrological services, wetlands are also valued for their cultural, educational and recreational importance (Boyer & Polasky, 2004).

The function of a wetland is based on how it is classified; however, wetlands are generally multi-functional and multi-value ecosystems (Finlayson & Van der Valk, 1995; Brander et al. 2006). Wetlands function as part of a landscape regardless of the presence or absence of humans, yet the perceived value of wetlands is based on ecological processes determined by human perceptions and scale (Mitsch & Gosselink, 1993). The function of a wetland, and ultimately its value is based on several factors: wetland type or classification, scale of analysis, location in landscape, interactions with other ecosystems, pressure from human populations, climate, vegetation and watershed characteristics, and the water balance (Mitsh & Gosselink, 2000; Brander et al. 2006). General wetland function and perceived value can be grouped into three categories (Table 2).

Table 2: General description of the functions and associated values of wetlands adapted from Wray & Bayley (2006).

	Function	Value
Physical	Groundwater recharge/discharge, flood water storage, erosion control, peat accumulation	Flood mitigation, base flow provider, water storage, sediment trapping, nutrient accumulation (e.g., Carbon)
Chemical	Toxicant and nutrient removal	Increased water quality
Biological	Wildlife/waterfowl habitat, biomass productivity	Breeding, nesting, diverse communities, recreational activities like hunting and fishing, rare plant species

In the last few decades, the economic value of wetland ecosystem services has been debated and widely studied (Woodward & Wui, 2000). In a review covering 33 studies, the range of estimates spanned $0.06 USD/acre to $22,050 USD/acre, even within the same ecosystem and study (Heimlich et al. 1998). The World Wildlife Federation (WWF) published the first comprehensive overview of the world's wetlands using 89 valuation studies and a database that covered 630,000 km^2. Conservative estimates from this study approximate a global value of $3.4 billion USD. However, using the RAMSAR Convention's world-wide area estimate of 12.8

million km^2, the global worth of wetlands is predicted to be $70 billion USD (WWF, 2017). The WWF evaluation of the mean areal unit value of wetlands is $5,582 USD/acre/year (Table 3).

A significant challenge in the valuation of wetlands is the inadequate understanding of the link between changes in wetland structure and function with wetland value (He et al. 2015). Throughout the literature, valuation techniques, the services being considered, wetland type and location vary considerably and often result in wetlands being undervalued (Brander et al .2005; de Groot et al. 2012). Paradoxically, the unit areal value of some wetlands rises with anthropogenic disturbance due to increasing scarcity, yet the functions and services they provide can be completely overwhelmed by the same disturbance, and ultimately decrease their economic value (Mitsch & Gosselink, 2000). For example, WWF (2004) estimated the economic value of wetlands per geographic area and found Asian wetlands to be the most valuable (i.e., $1.8 billion/year; Schuyt & Brander, 2004). However, the scarcity of wetlands in Asia is a result of development pressure, drainage, and land-use changes that have resulted in complete loss of wetland function in many areas (Gopal, 2013). Wetlands function best in a landscape as spatially distributed complexes; however, their economic value increases when there are fewer wetlands in the vicinity (de Groot et al. 2012). Therefore, an issue with quantifying the value of wetlands and other ecosystems in a single value (e.g., per ha) is that it can be used to argue for the replacement of one system over another when the landscape view is not considered (Mitsch & Gosselink, 2000).

Table 3: Average wetland value per acre in USD for the year 2000. Adapted from (WWF, 2017).

Wetland Function	**Median Wetland Economic Value (US/acre/year, 2000)**
Flood Control	1,146
Rec. Fishing	924
Amenity/Recreation	1,215
Water Filtering	711
Biodiversity	529
Habitat Nursery	496
Recreational Hunting	304
Water Supply	111
Materials	111
Fuel Wood	35
Total	**$5,582**

Wetlands are also valuable to other species and act as a storehouse for biodiversity (Keddy 2010). The productive nature of wetland ecosystems generates thousands of species of plants, such as mosses, grasses, shrubs, trees and flowers, which provide habitat for mammals, birds, fish and invertebrates, and serve as nurseries for these species and many others (Ducks Unlimited Canada, 2008). For instance, it is estimated that 20% of the approximately 9000 species of birds in the world are wetland dependent (Wetlands International, 2018). In Canada, wetlands are home to one-third of the threatened and endangered species and are important to waterbirds for nesting, breeding and feeding (Environment Canada, 2004). Further, wetlands in Canada generate an estimated $5-10 billion CND dollars annually in economic value through the hydrological, biological and chemical services they provide as well as recreational uses (Environment Canada, 2004).

Despite the increasing awareness of the importance of wetlands and attempts to quantify their value, over 50% of the world's wetlands have been lost (Mitsch & Gosselink 2000) and a growing number of studies demonstrate that wetland area and quality continue to decline in Canada and around the world (e.g., see Ramsar, 2018). Due to this rapid decline, the ecosystem services that wetlands provide are increasingly compromised, which has led to an emphasis on timely assessments for better management of wetland ecosystems.

3. Wetland Management

Wetland management has never been as difficult as it is today within our highly fragmented and human-modified landscapes (Zelder & Kercher, 2005; Euliss et al. 2008; Erwin, 2009). Wetland science and management developed in response to widespread wetland loss and concerns about the negative impacts on the biotic species, particularly migrating birds (Euliss et al. 2008). Decades later, the continued conversion of natural landscapes for urban and agricultural purposes has now compromised even the land dedicated to wetland conservation (Patenaude et al. 2015). The growing effects of anthropogenic disturbances on wetland landscapes has rendered many of the traditional wetland management techniques ineffective as they were based on an incomplete understanding of landscape processes and/or social influences (Euliss et al. 2008). Recently, decreased wetland cover was found to be more detrimental than urbanization to overall wetland quality and the authors suggest that policies aimed at wetland management should be applied to entire landscapes (Patenaude et al. 2015). New perspectives need to be developed not only to create new wetlands, but to manage the long-term sustainability of crucial wetland ecosystems within the constraints of increasingly disturbed landscapes and a changing climate.

3.1 Wetland disturbance

Wetland disturbance can be defined as any phenomena or process that alters the function of a wetland or wetland complex outside of normal or natural ranges (Detenbeck et al. 1999). This includes both natural and anthropogenic events that cause changes to the entire ecosystem, the community or population structure, and/or deviations in the chemical and physical environment (Picket & White, 1985). Generally, disturbance to a wetland ecosystem can be grouped in three categories 1) complete loss of wetland and function, 2) direct disturbance, and 3) indirect disturbance to wetlands (Wray & Bayley, 2006).

The typical response of wetland systems to disturbance commonly results in changes to biotic composition, diversity and/or productivity, and leads to decreased water quality, while the impacts of larger (i.e., landscape-level) disturbances often manifest as changes in the overall hydrology and affect key functions such as: sedimentation, biomass removal, water retention time and nutrient enrichment (Detenbeck et al. 1999). Both local and landscape-level disturbances can increase the susceptibility of wetlands to invasive species or the dominance of native species (e.g., *Typha* spp) (Galatowitsch et al. 2000). Although natural variation is quite common and can have dramatic effects on wetlands and their appearance, healthy wetlands tend to have an abundance of hydrophytes (i.e., emergent and submerged) that sequester nutrients and decrease turbidity, while disturbed wetlands generally have little vegetation, increased turbidity and higher nutrient loads (Chow-Fraser, 1999).

3.2 Monitoring wetlands

Timely and consistent wetland monitoring and assessment is critical for managing and protecting wetland resources (EPA, 2018). To successfully monitor wetlands requires creating an inventory, establishing baseline or reference conditions, detecting change, and analyzing trends (NRC, 2016; EPA, 2018). A variety of indicators have been used to monitor and measure wetlands, which can be grouped into three broad categories: biological (e.g., macro-invertebrates), chemical, (e.g., dissolved oxygen), and physical (e.g., hydrologic and topographic) (Adamus, 1992). While biological and chemical indicators are the most common parameters for monitoring wetlands and assessing health, they are not the most practical due to the accessibility, cost and time constraints of field work (Wray & Bayley, 2006). Recent technological advancements such as, satellites and other remote sensing technologies (i.e., planes, remotely piloted aircraft) have shown to be powerful tools for monitoring and identifying changes in wetlands on a regional and national scale (NRC, 2016). These technologies provide several advantages for measuring and monitoring wetland ecosystems: the ability to provide more timely and affordable data, increased accessibility to remote areas, the ability to analyze larger geographic areas, and quantify results using landscape metrics (Ozesmi & Bauer, 2002).

3.3 Man-made wetlands

Despite our best efforts, man-made wetlands often result in aquatic plant communities (Rooney & Bayley, 2011b) soil and water chemistry (Scholz & Lee, 2005) that differ from what is typically found in natural undisturbed wetlands. The creation of wetlands has been supported through various international agreements (e.g., RAMSAR Convention on Wetlands, 1993) and government policy (e.g., Alberta's Wetlands Policy, 2013). Generally, there are three types of man-made wetlands: constructed, restored and reclaimed.

Constructed wetlands are those that were created where no wetland previously existed. For example, wetlands are often constructed for water management purposes in urban areas (Dou et al. 2017; Sharley et al. 2017). Wetland restoration is defined as returning a disturbed wetland (e.g., drained) back to its original form and function (Zelder, 2000; Erwin, 2009). Typically, this involves one of several goals: to restore function or biotic communities, create entirely new synthetic communities or re-establish them on the same site where they were destroyed (Keddy, 2009). The effectiveness of wetland restoration is often debateable as some level of "success" is required to justify the high costs associated with it (Zedler, 2000). Further, due to the complex interrelationships between biotic and abiotic components, wetland function and biodiversity are unlikely to be maximized in the same restored wetland. For instance, restored wetlands are often designed to return nutrient removal functions, but this requires eutrophic conditions. Eutrophic conditions in wetlands often lead to a vegetation community that is dominated by a single hydrophyte species (e.g., *Typha* spp.), which limits the overall biodiversity and productivity of that wetland (Zelder, 2000).

In contrast, reclamation typically involves recreating or modifying a disturbed environment to achieve some biotic target, such as making a disturbed landscape appear natural, achieve some level of sustainability, and integration with surrounding landscape (GOA 2016; Rooney et al. 2015). Reclamation is a general term with legal definitions that varying across governments and countries (Harrington, 1999) and wetland reclamation may involve the restoration or construction of wetlands at small or very large spatial scales (Rousseau et al. 2008; Rooney & Bayley, 2011b). Research on wetland reclamation has demonstrated that plant diversity is affected by the surrounding landscape, with direct effects having the most influence within 500 m and indirect effects having more influence on plant diversity at 1500 m or greater, signifying the mechanisms that contribute plant diversity in reclaimed wetlands are dependant on the spatial extent of the surrounding landscape (Rooney & Bayley, 2011b). For example, if impediments in the surrounding landscape limit the propagation of seeds (i.e., direct effects), the flux of nutrients, and/or energy or materials (i.e., indirect effects) between wetlands, then man-made wetlands will likely fail to meet objectives and support diversity on levels comparable to natural wetlands (Galatowitsch & van der Valk, 1996; Rooney & Bayley, 2011b). The importance of ecological and hydrological connectivity to the surrounding landscape suggest that a landscape-level approach may be more successful than a localized one when trying to create wetlands that are integrated with the surrounding landscape and are self-sustainable (Zelder, 2003).

In Canada, approximately 70% of wetlands have been destroyed or degraded and restoration and reclamation success has been limited by cost and time constraints (Ducks Unlimited Canada, 2008). For example, in the province of Alberta, more than 813 km^2 of the western Boreal has been disturbed by oil sands mining (Rooney et al. 2015) with only about 60 km^2 being permanently reclaimed, which means the restoration work has been completed but it has yet to be certified by the province (Rooney & Bayley, 2011a; COSIA, 2018). Reclamation can take decades and the only land to be certified by the province as reclaimed is a 104-hectare plot known as Gateway Hill, that Syncrude began working on in the 1980's and which contains no wetlands (Syncrude, 2018). More recently, both Syncrude and Suncor have attempted to reclaim fens, which have been hypothesized to be an achievable reclamation target in post-disturbance landscapes in northern Alberta (Price et al. 2010); however, doubts remain about the long-term success of such projects given the salinity of the soil in these post-disturbance landscapes (Purdy et al. 2015) and the cost associated with reclamation (Foote, 2012).

Researchers familiar with these projects conservatively estimate $4.3 to $12.9 billon will be required to reclaim all the disturbed wetlands in the oil sands region of Alberta (Foote, 2012), and argue it may not be cost-effective to restore the landscape to its original form and function, particularity considering decades of mismanagement by the Alberta government and volatile oil prices (Lemphers et al. 2010). Further, a study that examined an area approximately 150 km^2 in northeast Alberta found significant issues existed with landscapes where the reclamation work had been completed, but not yet certified. Major concerns were listed as: reduced biomass and amount of native plants, homogeneity of the landscape, and elevated concentrations of salts and contaminants in the soil, which have been shown to bioaccumulate in the environment (Timoney, 2015).

4. Wetlands in Alberta

The landscape in Alberta is peppered by millions of wetlands that collectively cover approximately 21% of the land area in the province, 90% of which are peatlands (NWWG, 1988; Conly & Van Der Kamp, 2001.) Not only do these wetlands provide critical habitat to many protected species, they host millions of migratory and breeding birds each year. For example, the northern leopard frog, trumpeter swan, whooping crane, piping plover, and woodland caribou are all species at risk that rely on these wetlands (GOA, 2016). Further, they have significant cultural importance for First Nations and Metis as a large portion of the land in Alberta used by these communities contains wetlands (AEP, 2018).

Topographically, wetlands in Alberta are often surrounded by upland at the local-level and are frequently referred to as geographically isolated wetlands (GIW) (Mushet et al. 2015). GIWs represent most of the wetlands in the North American landscape and embody many different wetland types. The absence of year-round surface water connectivity has led to GIWs being falsely interpreted as “functionally isolated” (Tiner, 2003; Cohen, et al. 2016). Despite their slow and intermittent surface connectivity, they provide important biological, hydrological and biogeochemical exchanges with neighbouring waterbodies via groundwater, which allows for the dispersal of energy, materials and organisms (Tiner, 2003; Mushet et al. 2015). GIWs form complexes with other waterbodies to create temporal and spatial heterogeneity with respect to the timing, magnitude and flow of water, which connects a larger hydrological network

(Cohen, et al. 2016). In the semi-arid regions of Alberta, these wetlands play a critical role in ecological processes due to the groundwater recharge that occurs from the snow-melt they retain (Price et al. 2005). However, many of these wetlands are in topographically low areas, so they are particularly vulnerable to the impacts of disturbance (Adamus, 1992).

The government of Alberta estimates that approximately two-thirds of wetlands have been lost due to increased urbanization, resource extraction and agriculture (AEP, 2013; Main et al. 2014). Alberta has several policies aimed at managing wetland ecosystems more efficiently (e.g., the Water Act, the Alberta Wetland Policy, the Environmental Protection and Enhancement Act, and the Public Lands Act; AEP, 2013), and in 2003 released their *Water for Life Action Plan*, which includes objectives to better improve the understanding of wetland ecosystems and the ability to assess wetland reclamation projects (AWP, 2017). Further, the Conservation and Reclamation Regulation within the Environmental Protection and Enhancement Act (EPEA), requires reclaimed wetlands to be 1) naturally appearing 2) self-sustaining and 3) integrated with the surrounding landscape (GOA, 2016), but currently, these polices are void of any design criteria for reclaiming entire wetland landscapes (AESRD, 2013).

4.1 Alberta wetland classifications and inventories

The Alberta Wetland Classification System (AWCS) encompasses information from existing wetland classifications such as the CWCS and Stewart and Kantrud (1971) to create a tailor-made system that is applicable to the entire province (GOA, 2015). Overall, wetlands in the AWCS are separated into two broad groups and five classes: peatlands (i.e., bogs and fens) and mineral wetlands (i.e., marshes, fens and shallow open-water), which align with the CWCS. The wetland classes are then sub-divided into types based on duration of inundation. The Stewart and Kantrud (1971) classification system is incorporated at the sub-level of wetlands types (Table 4).

Numerous wetland inventories also exist in Alberta: the Alberta Wetland Inventory (GOA, 2015), which is used extensively in the Boreal region, the Grassland Vegetation Inventory for lentic ecosystems in southern Alberta, and the Ducks Unlimited Boreal Plains Ecozone Classification (Smith et al. 2007). The Alberta Merged Wetland Inventory (AMWI) was released by Alberta Environment and Parks in 2014 to expand upon the Canadian Wetland Inventory (2002) and depicts wetlands in Alberta from 1998 to 2015. The AMWI is a generalized, merged product of 33 inventories that used a collection of different data sources for different years, various capture techniques and different classifications (AEP, 2018).

4.2 Natural regions

Natural regions are the largest ecological units used for land-cover mapping in Alberta. They are delineated geographically based on vegetation, landscape pattern, soil and physiological features, which reflect the influence of geology, topography and climate (Downing & Pettapiece, 2006). While there are six Natural Regions and 21 subregions in Alberta, this research is focused on the Prairie Grassland, Aspen Parkland and southern Boreal Mixedwood regions (Table 5).

Due to its position, Canada is expected to experience greater rates of warming than many other countries in the world (Erwin, 2009) and a northward shift of natural regions is expected (i.e., Grassland and Parkland regions are expected to shift north, replacing much of Boreal region) (Lemieux & Scott, 2005; Schneider et al. 2015). Within Canada, the prairies are expected to see the greatest changes in climate compared to other regions (Erwin, 2009). Therefore, it may be more useful for modeling and creating wetlands in the future, to understand the landscape patterns that contribute to creating and maintaining various wetlands types within their respective natural region. In doing so, a baseline or reference condition for wetlands based on the natural regions can be created that could aid in managing and creating wetland landscapes in the future and increase our ability to meet policy objectives.

Table 4: Wetland classes, forms, types in the Alberta Wetlands Classification System with classification codes for mapping in brackets. Adapted from Alberta Wetland Policy (2017).

<table>
<tr><th rowspan="2">Class</th><th rowspan="2">Form</th><th colspan="3">Types</th></tr>
<tr><th>Salinity</th><th>Hydroperiod</th><th>Acidity-alkalinity</th></tr>
<tr><td>Bog (B)</td><td>Wooded, Coniferous (Wc)</td><td>Freshwater (f)</td><td>-</td><td>Acidic (a)</td></tr>
<tr><td rowspan="3">Fen (F)</td><td rowspan="3">Wooded, Coniferous (Wc), Shrubby(S), Graminoid(G)</td><td>Freshwater (f)</td><td>-</td><td>Poor (p)</td></tr>
<tr><td>Freshwater (f)</td><td>-</td><td>Moderate-rich (mr)</td></tr>
<tr><td>Freshwater (f) to slightly brackish (sb)</td><td>-</td><td>Extreme-rich (er)</td></tr>
<tr><td rowspan="3">Marsh (M)</td><td rowspan="3">Graminoid(G)</td><td>Freshwater (f) to slightly brackish (sb)</td><td>Temporary (II)</td><td>-</td></tr>
<tr><td>Freshwater (f) to moderately brackish (mb)</td><td>Seasonal (III)</td><td>-</td></tr>
<tr><td>Freshwater (f) to brackish (b)</td><td>Semi-permanent (IV)</td><td>-</td></tr>
<tr><td rowspan="4">Shallow Open-water (W)</td><td rowspan="3">Submersed and/or floating aquatic vegetation (A) or bare (B)</td><td>Freshwater (f) to moderately brackish (mb)</td><td>Seasonal (III)</td><td>-</td></tr>
<tr><td>Freshwater (f) to sub-saline (ss)</td><td>Semi-permanent (IV)</td><td>-</td></tr>
<tr><td>Slightly brackish (sb) to sub-saline (ss)</td><td>Permanent (V)</td><td>-</td></tr>
<tr><td>(A)</td><td>Saline (s)</td><td>Intermittent (VI)</td><td>-</td></tr>
<tr><td rowspan="3">Swamps (S)*</td><td rowspan="3">Wooded, Coniferous (Wc), Shrubby(S), Wooded, mixedwood (Wm), Wooded, Deciduous (Wd)</td><td>Freshwater (f) to slightly brackish (sb)</td><td>Temporary (II)</td><td>-</td></tr>
<tr><td>Freshwater (f) to slightly brackish (sb)</td><td>Seasonal (III)</td><td>-</td></tr>
<tr><td>Moderately Brackish (mb) to sub-saline (ss)</td><td>Seasonal (III)</td><td>-</td></tr>
</table>

* swamp types are not applicable to wooden swamps due to lack of information

** Roman numerals are equivalent to wetland classes by Stewart and Kantrud

Table 5: Natural regions covered in this study. Adapted from Downing & Pettapiece (2006).

Natural Region	Subregions
Southern Boreal	Dry Mixedwood Central Mixedwood
Parkland	Foothills Parkland Central Parkland Peace River Parkland
Prairie Grassland	Dry Mixedgrass Mixedgrass Northern Fescue Foothills Fescue

4.2.1 Prairie Grassland region

The Prairie Grassland region of Alberta is characterized by numerous small depressional wetlands (i.e., potholes) that were carved out by receding glaciers approximately 10,000 years ago (Winter, 1989). The Prairie Grassland region covers most of the south-eastern part of the province and extends into the prairie pothole region (PPR) ecozone, which spreads into the U.S. (Wray & Bayley, 2006). While this region is exclusively a grassland ecozone, the PPR extends into the Aspen Parkland region as well. The Grassland region has four subregions: dry mixed grass, mixed grass, foothills fescue and northern fescue (Downing & Pettapiece, 2006). The average annual temperature in this region is 4 °C, but seasonal temperatures can range widely from -40°C to +40°C and exhibit alternating periods of deluges and droughts (Winter, 1989; Diaz, 1986). Such fluctuations in temperature and precipitation results in dynamic changes in vegetation and productivity. For example, during a wet phase, wetlands form into larger ponds that reduce the amount of emergent vegetation and productivity seen in wetlands during drier periods (Wray & Bayley, 2006). Due to these extremes, prairie grassland wetlands have a complex association with groundwater recharge and discharge that depends on the underlying geology and climatic variations (Wray & Bayley, 2006). The overall wetland water balance in this region is controlled by snowdrift and snowmelt runoff from the surrounding uplands, evapotranspiration, the occasional “fill-spill” that connects to other wetlands, and through groundwater exchange (NWWG, 1988).

In the Grassland natural region, estimated wetland loss is approximately 35-40%. Most of these wetlands were lost through drainage for agriculture and over 90% of the remaining wetlands are adversely affected by urbanization (AEP, 1997; Wray & Bayley, 2006). Seasonal and temporary wetlands are abundant in this region. They are extremely dynamic in nature and are the least protected wetlands and at the greatest risk for loss due to urbanization and agriculture (Naugle et al. 2001; Wray & Bayley, 2006; Serran & Creed, 2015).

4.2.2 Aspen Parkland

The Aspen Parkland region of Alberta is divided into three subregions: Central Parkland, Peace River Parkland, and Foothills Parkland (Downing & Pettapiece, 2006). The Aspen Parkland is a transitional zone between southern prairie grasslands and Boreal forest to the north, covering approximately 12% of the province's land surface (Strong & Leggat, 1981). The southern part of this region is characterized by grasses with aspen that grow in moist depressions, while the northern part of this region is marked by continuous forest. The mean annual temperature of this region is 2°C (Downing & Pettapiece, 2006) and is generally wetter and colder than the prairie Grassland region in the south (Environment Canada, 2004). Wetlands here are mostly marsh and shallow open-water that are like those found in the prairie Grassland region but tend to be more permanent due to wetter climate (NWWG, 1988). The Parkland region is the most densely populated region and is highly fragmented due to agriculture (Wray & Bayley, 2006).

4.2.3 Boreal Mixedwood region

The Boreal Mixedwood region of Alberta is the largest ecozone in the province, which covers approximately 43% of the land surface (Downing & Pettapiece, 2006). The mixedwood region is located between the Aspen Parkland and the Boreal forest to the north and is divided into three subregions: dry mixedwood, moist mixedwood and wet mixedwood, which are based on summer precipitation (Strong & Leggat, 1981). This region has a mean annual temperature of 0.7 °C and is characterized by brief and cool summers and long cold winters. The topography can be described as rolling terrain, lacustrine and moraine deposits, with a few areas of outwash and sandy dunes and is a region subject to frequent forest fire (Wray & Bayley, 2006). Peatlands (i.e., fens and bogs) are the dominate wetland types in this region (Moore, 2002), but marsh wetlands are commonly found on the fringes of open-water and fen wetlands. Marshes that surround depressional ponds tend to accumulate significant peat deposits as well (Bayley & Mewhort, 2004). The dominate disturbance in this region is resource extraction with some encroaching agriculture (Downing & Pettapiece, 2006; Wray & Bayley, 2006).

The wetland-rich Boreal region is home to one of the world's largest mega-projects (i.e., oil sands mining) (Rooney et al. 2015), which will result in the largest wetland reclamation effort in Canadian history (Rooney & Bayley, 2011a). To increase the success of industry-led landscape reclamation efforts in Alberta, a landscape-level reference condition approach that considers the uniqueness of each wetland type and natural region may be beneficial.

5. Oil Extraction in Alberta

In-situ oil extraction occurs mostly in the Boreal region and represents approximately 80% of all the mining in Alberta (CAPP, 2015). In-situ oil extraction requires an extensive infrastructure network and the total disturbance footprint encompasses: well pads, gravel pits, steam generators, bitumen treatment plants, water treatment plants, and worker camps (Evans et al. 2017). Utilizing satellite imagery, the scale of reclamation for these sites was estimated to have an average spatial extent of 1 km^2, indicating a landscape-level analysis will be required to reclaim these sites successfully upon closure (Evans et al. 2017). The reclamation of such large-scale mega-projects comes at considerable financial cost due to the differences in topography between the disturbed area and the surrounding landscape, creating a total reclamation debt that was estimated to be $15 billion in 2012 (Lemphers et al. 2010). Accounting for an average yearly inflation rate of 1.35%, that is approximately $17 billion today (Beard et al. 2018). Furthermore, extraction can occur over decades and these sites will experience significant climate change that will alter the vegetation and hydrology of the landscape (Richens, 2010). Current reclamation closure plans included in permit applications do not consider future climate change (Rooney et al. 2015). While some progress that has been made in assessing aquatic systems (e.g., ABWRET-E, ABWRET-A; Bruneau, 2017; Creed et al. 2018), and it is required that industry use the province's wetland reclamation guideline (GOA, 2016), current design and evaluation criteria for oil sands reclamation certification is focused on individual habitat patches and do not consider the cumulative long-term sustainability of patches at the landscape-level (AESRD, 2013; Timoney, 2015). This is especially important for wetlands, which occupy the basins of watersheds and rely on hydrological processes for wetland creation (Adamus, 1999).

5.1 The role of open-water wetlands in oil sands reclamation

Reclamation is an ongoing process during the life of an in-situ oil extraction project. Operators in the oil sands must develop a plan to reclaim the land and have it approved by government as part of the project approval process (Rooney et al. 2015). Companies then apply for a government closure permit once reclamation has been completed, the vegetation is mature, the landscape is self-sustaining, and the land can be returned to the Crown for public use (Canadian Oil Sands, 2018). An economically viable solution for these companies is to fill open quarries and pits with water, thereby creating an open-water wetland system, which is reflected in closure permit plans (Globe and Mail, 2012; OSAR, 2013). A challenge in reclaiming wetland landscapes, is that large-scale disturbances typically sever important hydrological connections and substantially alter the topography that sustains the water balance necessary for wetland creation (Foote, 2012). Further, fens are the dominate wetland type in the oil sands region (Price et al. 2012) and replacing these wetland-rich landscapes with upland forest and novel open-water wetlands will alter the function of these landscapes. Open water increases evapotranspiration, which may accelerate open-water wetland loss under future climate scenarios that predict increases temperature and sporadic precipitation (Winter, 2000; Sánchez-Carrillo et al. 2004). The effects of replacing native wetland types with novel open-water wetlands or how sustainable they would be under future climate scenarios are not considered in closure permit plans. Lastly, the absence of criteria for the certification of wetlands for landscape-level reclamation projects (AESRD, 2013) interferes with our ability to assess and monitor them.

6. Thesis Overview

Over the coming century, hundreds of wetlands are expected to be reclaimed in the oil sands region of Alberta, Canada (CEMA, 2014). Furthermore, the life span of such mega-projects will necessitate the consideration of climate change when attempting to create wetlands that meet the policy objectives outlined by the Government of Alberta (Rooney et al. 2015). While industry is required to follow the government regulatory guidelines for the reclamation of wetlands, the lack of design criteria within those guidelines for landscape-level reclamation is a significant gap in policy. The role that large permanent open-water wetlands will play in industry-led oil sands reclamation, requires an investigation of the composition and configuration of permanent open-water wetlands along disturbance levels and across the natural regions, to ensure a hydrological setting that can sustain these wetlands under future climate scenarios. Further, the number of expected permanent open-water wetlands to be created in areas where they are not typically found will require updating the current AWCS/CWCS to reflect this new genre of man-made wetlands in future inventories to ensure the proper management of these novel wetland landscapes.

Based on a review of literature concerning wetlands, their management, and the reclamation of wetland landscapes in oil sands mining, two significant overarching questions remain: what is the landscape structure associated with permanent open-water wetland-rich landscapes and how does that landscape structure vary with disturbance and across natural regions? To answer these questions, the distribution of permanent open-water wetland-rich landscapes are assessed across three natural regions in Alberta, Canada, and a method for quantifying the composition and configuration of those landscapes is outlined. This will involve 1) the creation of thousands of randomly generated 1 km^2 open-water wetland-rich landscapes across the study area 2) examining the frequency and distribution of permanent open-water wetlands and the level of disturbance in those landscapes 3) applying a set of parsimonious landscape metrics to the samples and 4) analyzing them statistically. The results from the aforementioned steps will provide insights that will increase our understanding of how permanent open-water wetlands respond to varying levels of anthropogenic disturbance within different natural regions and inform as to how wetland landscape metrics may be utilized to facilitate landscape-level reclamation. Further, this research may provide a baseline for analyzing the landscape patterns of permanent open-water wetland-rich landscapes in the Boreal, Parkland and Grassland regions, so that we can better reclaim these wetlands in the future.

Chapter 2—Quantifying Land-Cover Patterns Along Permanent Open-water and Disturbance Levels for Landscape-level Wetland Reclamation

1. Introduction

Persistent anthropogenic disturbances have affected the stability of crucial ecosystems and the services they provide on a global scale (Foley et al. 2005; MacDougall et al. 2013). Traditionally, in the field of ecology the concept of disturbance was void of anthropogenic influence but could broadly be defined as "any relatively discrete event in time that disrupts ecosystems, community or population structure, and changes resource pools, substrate availability, or the physical environment" (Pickett & White, 1985, pg. 371). More recently, the definition of disturbance in ecology has broadened to include anthropogenic processes such as: urbanization, resource extraction, and agriculture, which now affect most of the world's ecosystems to some extent (Hobbs et al. 2006). An increasing awareness of the effects from anthropogenic disturbances on the world's ecosystems has led to an emphasis on managing landscapes more efficiently, as well as efforts to recreate disturbed or degraded ones (Harris et al. 2006).

While restoring a landscape (i.e., returning a landscape back to its original state) is ideal, it is more common that disturbed landscapes are reclaimed (i.e., designed to be naturally appearing and self-sustaining), particularly in areas where resource extraction occurred (SMCRA, 1977; SER, 2004; GOA, 2016). Resource mega-projects (e.g., oil sands mining, diamond mining, mountaintop removal) present a unique reclamation challenge due to the massive spatial extents and timeframes of their operations (Johnson & Miyanishi, 2008; Rooney et al. 2015). Entire ecosystems, landforms and aquatic systems must be recreated and integrated with the surrounding landscapes. In addition to the engineering challenge this represents, meeting the goals of key stakeholders (e.g., government, industry, and citizens) is problematic (Rooney & Bayley, 2010) as post-disturbance landscape reclamation is expensive (Lemphers et al. 2010) and often creates novel ecosystems with different biotic assemblages and abiotic components that alter the hydrological connectivity of the landscape (Devito et al. 2012; Laarmann et al. 2015). Recreating aquatic systems further complicates large-scale reclamation projects due to the regulatory policies that often govern protected waterbodies, such as streams and wetlands (Gerlak, 2005).

Wetlands are valuable at-risk ecosystems (Woodward & Wui, 2001), which have declined by 50% world-wide in the last century (Mitsch & Gosselink, 2000; Fraser & Keddy, 2005). Wetlands are multi-functional systems that can provide a range of valued ecosystem services (Finlayson & Van der Valk, 1995; Brander et al. 2006) that is disproportionate to the spatial extent they occupy both nationally and globally (Zedler & Kercher, 2005). For example, wetlands may help to mitigate floods, filter water (Woodward & Wui, 2001), store carbon (Euliss et al. 2006), support biodiversity, and provide habitat to rare and at-risk species (Keddy, 2010). In addition to their hydrological and ecological functions, wetlands have significant cultural, recreational and educational value (Boyer & Polasky, 2004). Due to this economic, ecological, and cultural importance, wetlands are often the focus of many conservation, restoration, and reclamation efforts (Mitsch & Wilson, 1996; Chimner et al. 2017). Several countries and regions have enacted polices to ensure the protection of existing wetlands and regulate their creation (e.g., the RAMSAR International Convention on Wetlands, the European Union's Water Framework Directive, Alberta's Wetland Policy).

With an extensive history of mega-projects (Maxwell et al. 1997; Jergeas, 2008) and reclamation efforts (Scheider et al. 1975; Powter et al. 2012; Rooney et al. 2015), Canada, and more specifically the province of Alberta, is currently undertaking significant wetland reclamation. Urbanization, agriculture and resource extraction have led to a significant decline in wetlands since the late 1800s (Dahl & Johnson, 1991; Euliss & Mushet, 1996). The government of Alberta estimates that two thirds of its wetlands have already been lost and concedes that it does not fully understand the losses still occurring (AWC, 2008; AEP, 2013). For example, oil sands extraction in the Boreal region has led to 813 km^2 of wetland-rich landscape being disturbed — a number expected to increase significantly in the future (Rooney et al. 2011; Rooney et al. 2015). Over the next century, hundreds of wetlands are expected to be reclaimed in this region alone (CEMA, 2014), representing the largest wetland reclamation project in Canadian history (Rooney & Bayley, 2010). In Alberta, a closure plan describing reclamation plans is required to obtain a lease for public land (Rooney & Bayley, 2010). Legal definitions of reclamation vary across governments and countries, but regulations in Alberta require disturbed lands to be reclaimed upon mine closure and returned to a naturally appearing, self-sustaining state of equivalent land capability (Kompanizare et al. 2018; Alam et al. 2018). Many of these closure plans include designs to fill open quarries and pits with water, thereby creating a permanent open-water wetland ecosystem (Rooney & Bayley, 2010; OSAR, 2013). However, replacing native peatland-rich landscapes (Wray & Bayley, 2006) with novel open-water landscapes may result in our failure to meet policy objectives under future climate scenarios, which predict increased temperatures (Schneider et al. 2015), sporadic precipitation (Kompanizare et al. 2018), and reduced snow pack depth and spring melt (Zhang et al. 2001; Pike et al. 2008). Further, it is unknown how these novel open-water wetland landscapes compare to natural landscapes or how they respond to various levels of disturbance.

The management and reclamation of wetlands in Alberta is governed by several polices: The Water Act, the Environmental Protection and Enhancement Act, the Public Lands, and the Alberta Wetland Policy (AWP, 2013). In 2003, Alberta released their *Water for Life Action Plan*, which includes objectives to better improve the understanding of wetland ecosystems and the ability to assess wetland reclamation projects (AEP, 2017). While some progress that has been made in assessing aquatic systems (e.g., ABWRET-E, ABWRET-A; Bruneau, 2017; Creed et al. 2018), and it is now a requirement that industry use the province's wetland reclamation guideline (GOA, 2016), the current design and evaluation criteria for oil sands reclamation certification is focused on individual habitat patches and does not consider the cumulative long-term sustainability of patches at the landscape-level (AESRD, 2013; Timoney, 2015). This is especially important for wetlands, which occupy the base of their catchments and rely on hydrological contributions from the surrounding upland, making them sensitive to changes and disturbances within that catchment (Adamus, 1999). This oversight in the government's wetland certification inhibits the evaluation of current and future reclamation activities happening at the landscape level (Rooney & Bayley, 2010; Timoney, 2015; AEP, 2017).

Further, the lifespan of many resource mega-projects necessitates the consideration of climate change when attempting to create wetland landscapes that are sustainable in the future and meet policy objectives (Rooney et al. 2015). For example, due to its position the Canadian prairies are expected to see significant increases in temperature (Erwin, 2009) and a northward shift of the natural regions are expected (i.e., Grassland and Parkland regions are predicted to shift north, replacing much of Boreal region; Lemieux & Scott, 2005; Schneider et al. 2015). These circumstances and the role industry will play in reclaiming wetland-rich landscapes, require the need to understand the relationship between land-cover and permanent open-water wetlands across natural regions for the creation of sustainable landscapes.

The search for and creation of guidelines and parameters to facilitate reclamation design and assessment for wetland landscapes remains a continued focus of academia, industry and regulators. The presented research furthers this effort by exploring the relationship between permanent open-water wetland-rich landscapes and disturbance, and seeks to answer the question: What is the landscape structure associated with permanent open-water wetland-rich landscapes and how does landscape structure vary with disturbance and across the natural regions? To answer these questions, we assess the distribution of permanent open-water wetland-rich landscapes across three natural regions in Alberta, Canada, and outline a method for quantifying the composition and configuration of these wetland-rich landscapes along levels of anthropogenic disturbance using a select group of independent metrics. This exploratory framework may aid in designing landscapes that include permanent open-water wetlands that can sustain the hydrological requirements for maintaining the wetlands, despite fluctuations in the water budget resulting from climate change.

2. Methods

2.1 Study area

Natural regions are the largest units in Alberta's ecological land classification system. These units are uniquely defined based on soils, vegetation, and physiographic features, and their distribution is the result of influences from topography, climate and geology (Downing & Pettapiece, 2006) (see Appendix 1 for a description of natural regions in the study area). The presented research covers the Grassland, most of the Parkland and southern Boreal regions and is constrained by two non-overlapping wetland inventories (Figure 2.1). In the southern Boreal, wetlands are mainly peatlands with disturbances primarily caused by forestry, oil and gas, coal mining, with small amounts of till cropping and subsistence farming (Downing & Pettapiece, 2006). In the Parkland and Grassland regions, wetlands are mainly shallow open water or temporary wetlands with major disturbances primarily caused by oil and gas, agriculture and urbanization (Downing & Pettapiece, 2006). It should be noted that while both inventories cover a portion of the Parkland, neither cover the central area of this region (Figure 2.1).

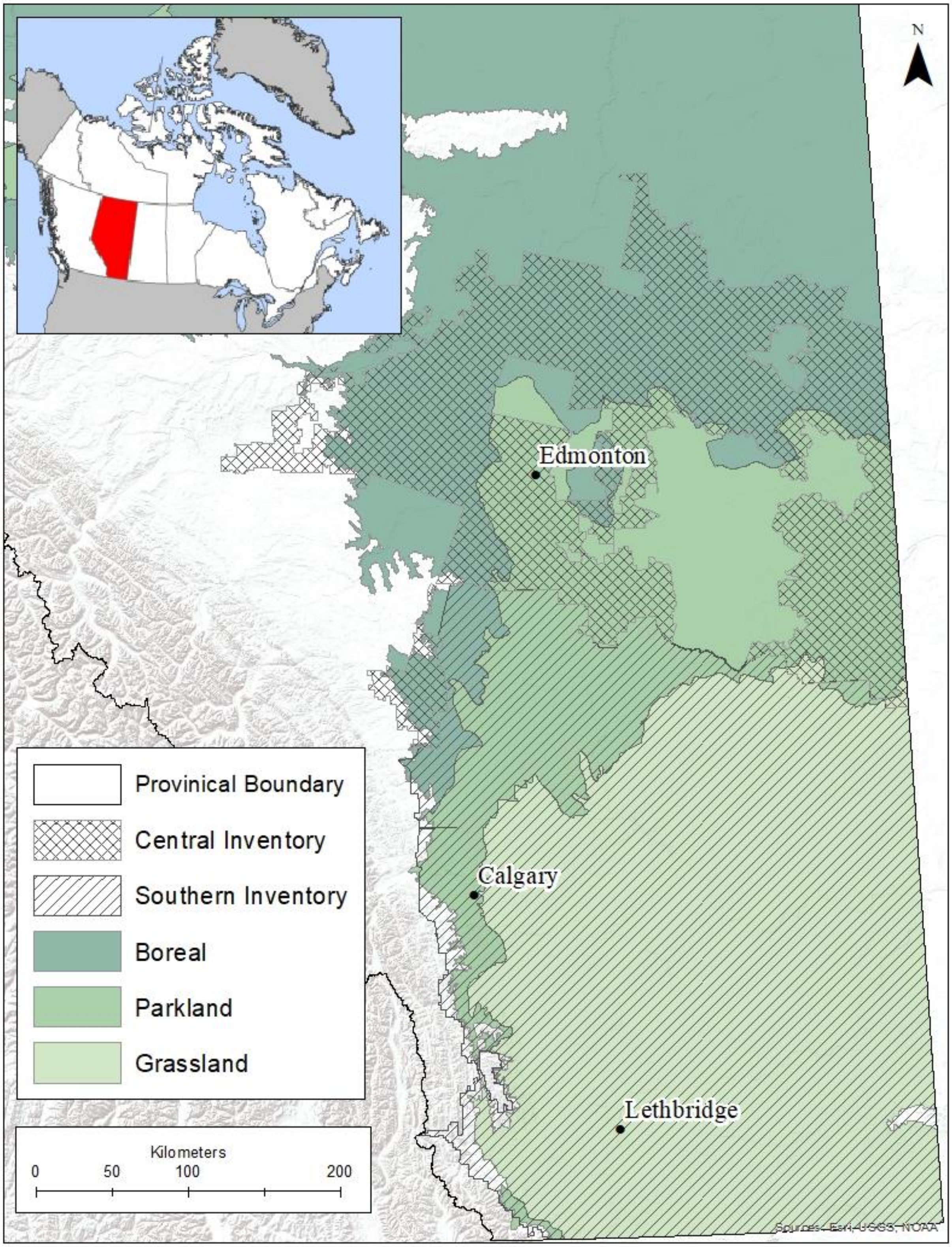


Figure 2. 1: The Central and Southern wetland inventories comprising the Grassland and portions of the Parkland and Boreal natural regions of Alberta, Canada. Within the inset, the province of Alberta is shown in red.

2.2 Data

Two wetland inventories (i.e., Central and Southern) were acquired from the Alberta Sustainable Resource Development (ASRD) and delineate lentic (i.e., still water) wetlands based on hydro-period: temporary, seasonal, semi-permanent, permanent open-water and alkali (Table 2.1) (ASRD, 2011). The Central and Southern inventories augment the Alberta Grassland Vegetation Inventory (GVI) classification system; however, different methods were employed in their development. The Central inventory covers a portion of the southern Boreal and north-west of the Parkland region (Figure 2.1) and was created with a combination SPOT 5 imagery (2006-2009), a 25 m resolution digital elevation model (DEM) and ancillary data (e.g., roads and hydrography) (ASRD, 2010). Uniform segments were created from the imagery with the size of each unit ranging from 0.001 ha to 2.0 ha. Object-based image classification was performed on the image segments to delineate land-cover types and a predictive ecosystem decision-tree model was used to produce the final wetland classes (ASRD, 2011). Classification accuracy was assessed using 100 random two-kilometre zones. Orthorectified and SPOT 5 imagery were manually interpreted for each zone and compared to the classification with observed accuracy at 83% (ASRD, 2011).

The Southern inventory was also created with SPOT 5 imagery but includes orthoimagery (2005-2006) and SPOT 4 images where cloud cover inhibited classification (Alberta Terrestrial Imaging Centre, 2009). Images were stacked temporally and classified with a support vector machine algorithm to identify wetland boundaries. A second classification was then performed to classify the wetlands by GVI classes (see Table 2.1) and the majority class identified over the image stack was assigned to represent that wetland feature. The minimum mapping units was 0.2 ha; therefore, wetlands less than 0.2 ha were excluded. The accuracy of the Southern inventory was ground-truthed in five township boundaries where wetlands were manually digitized and classified by aerial imagery with cell sizes between 0.5 m and 2.5 m. The range of accuracy among the five townships was 51%-68%, which is lower than the wetland classification accuracy of the Central inventory.

Table 2. 1: Description of lentic wetlands adapted from ASRD (2011).

GVI Code	Hydroperiod	Description
LenT	Temporary	Surface water retained only briefly after spring melting period. Low-Prairie and wet-meadow vegetation.
LenS	Seasonal	Surface water retained for more than three weeks. Lusher vegetation relative to Temporary wetlands due to higher water table.
LenSP	Semi-permanent	Surface water persists except in times of extreme drought. Emergent vegetation (e.g., cattails, bulrushes).
LenW	Open-water	Permanent open-water areas larger than 1ha.

Land-cover data were acquired from Agriculture and Agri-Food Canada (AAFC) annual inventory for 2009 to match with the ASRD wetland inventories. The AAFC dataset included 22 agriculture classes and nine non-agriculture classes (e.g., developed areas, vegetation type, wetlands, crop type) throughout the study area with a resolution of 30 m. This dataset was created from a combination of RADARSAT-2 and LANDSAT-8 imagery and classified using a decision tree method. An accuracy of ~90% was ground-truthed by crop-insurance companies. The 2009 dataset was aggregated into eight general land-cover classes: developed, agriculture, exposed, water, shrubland, wetland, grassland, and forest. Developed and agriculture land-cover classes were used to calculate anthropogenic disturbance. See Appendix 2 for complete land-cover list.

2.3 Utilizing landscape metrics for wetland reclamation

Landscape metrics are statistics that describe and quantify spatial characteristics of patches, classes of patches, or entire landscape mosaics (McGarigal & Marks, 1995). They have been used in a myriad of management applications ranging from analyzing land-cover change, to urban planning, and studies of biodiversity (Uuemaa, 2009). A variety of software packages exist to aid researchers in calculating landscape metrics (see Turner, 2005). Within such packages, landscape metrics are relatively simple to calculate and compare (Lausch & Herzog, 2002), making them ideal for government and industry partners to use when quantifying landscape features and for evaluating the success of large-scale reclamation efforts (Evans et al. 2017).

Landscape structure is the spatial pattern of the landscape and consists of two components: composition and configuration (McGarigal & Marks, 1995; Gustafson 1998). Composition is the non-spatial aspects and integration of patches (e.g., patch diversity, patch richness) (McGarigal, 2014), while configuration refers to the spatial arrangement, position, and orientation of elements within the landscape (McGarigal & Marks, 1995; Wei et al. 2017). Quantifying landscape structure is fundamental to understanding the interaction between landscape pattern and ecosystem function (Turner et al. 1989; Wei et al. 2017). For example. landscape configuration is critical to ensuring a hydrological setting that can deliver the water necessary to sustain a wetland (Ketcheson et al. 2016) and when combined with topography and climate, is a key determinate in wetland type (e.g., fen) (Mitsch & Gosselink, 2007). Furthermore, the composition of the surrounding landscape is known to influence aquatic species by affecting potential propagule sources and the flux of materials between wetlands (Rooney & Bayley, 2011), while the distance between wetlands can determine the presence and abundance of avian species (Fairbairn & Dinsmore, 2000). The importance of ecological and hydrological connectivity to the surrounding landscape requires consideration of the broader landscape structure when reclaiming wetland landscapes and may offer insight as to why individual reclaimed wetlands often have different aquatic and biotic communities than their natural counterparts (Rooney & Bayley, 2011).

2.4 Analysis

The analysis of open-water wetland-rich landscapes was a two-step process that involved generating random landscape samples and creating a subset for permanent open-water wetlands (LenW) for further statistical analysis (Figure 2.2). While much of this methodology was adapted from Evans et al. (2017), our study focused on permanent open-water wetland-rich landscapes to align with industry closure plans, increased the sample landscapes used from 3343 to 13, 676, removed 100% disturbed landscapes, tested for variance, and ensured the samples were spatially independent by applying a 1000 m minimum distance between landscapes (Kraft, 2016).

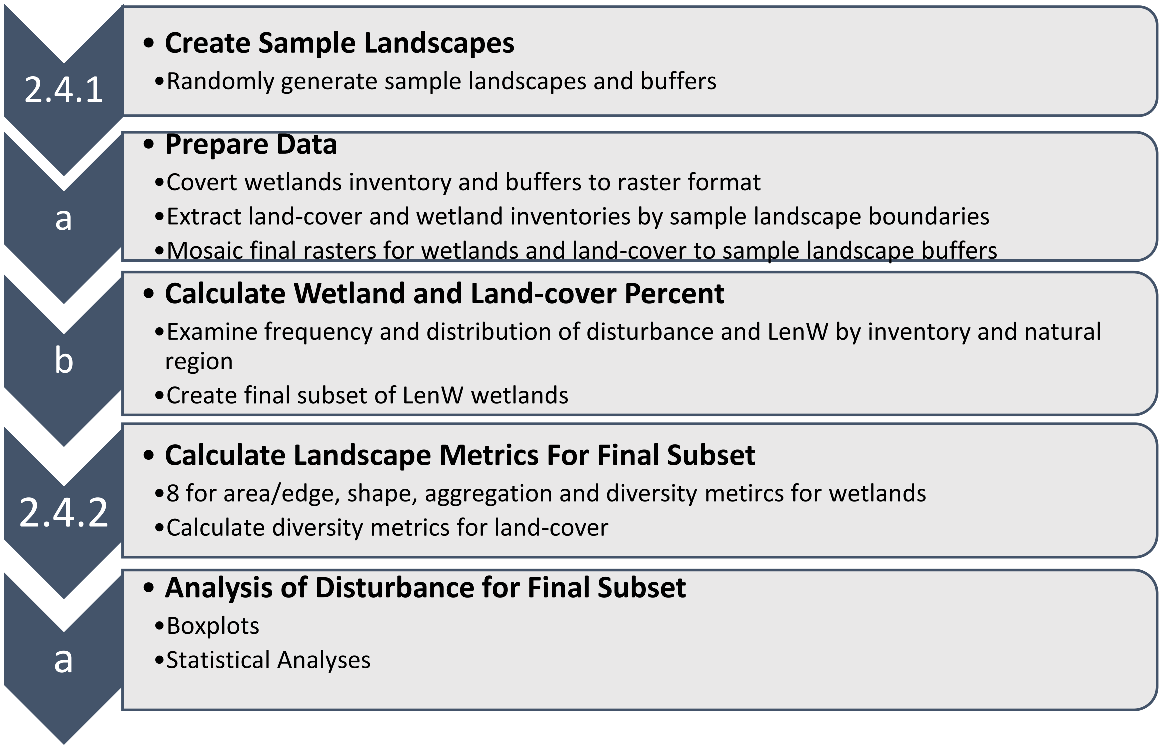


Figure 2. 2 Overview of analysis adapted from Evans et al. (2017).

2.4.1 Creating sample landscapes

Analysis was performed at a 'reclamation scale' that encompasses a disturbance footprint associated with in-situ oil extraction sites in Alberta, which was estimated to be 1 km^2 based on: the area disturbed by bitumen treatment plants, well pads, roads, gravel pits, steam generators, water treatment plants, and housing units for workers (Evans et al. 2017). A total of 13, 676 – 1 km^2 square sample landscapes were randomly generated throughout the study region (i.e., Central and Southern inventories) with a minimum allowed distance of 1000 m to ensure samples were spatially independent (Kraft, 2016). The wetland samples were converted to raster format for the calculation of metrics.

Percent land-cover and the percent of the landscape containing permanent open-water wetlands was calculated, and the results were interrogated by wetland inventory (Central and Southern) and natural region (Boreal, Parkland and Grassland), for a total of six datasets (Central All, Central Boreal, Parkland Boreal, Southern All, Southern Parkland, and Southern Grassland), through a series of frequency plots and boxplots based on percent of permanent open-water wetland (LenW) and disturbance level (Appendix 3). These datasets were then grouped in twenty-percent disturbance intervals (>0-20%, 21-40%, 41-60%, 61-80, 81-100%) for further analysis. Other methods of identifying breaks were explored (i.e., natural breaks with three, four, and five classes, equal breaks with five and ten classes and quantiles), but equal breaks with five classes in twenty percent intervals was chosen to make our results comparable across natural regions and inventories, to previous and on-going research (Evans et al. 2017; Branton 2018), and to increase the usability of the results by non-academics involved in wetland reclamation. The majority (82.82%) of all our sample wetland landscapes contained less than or equal to 20% permanent open-water wetlands (LenW). See Appendix 4 for boxplots of LenW in the sample landscapes.

Landscapes containing greater than 20% permanent open-water were randomly selected (n =30) and visually inspected using SPOT (2009) imagery. Among this sample, 93.33% were misclassified as they were part of a larger body of water (i.e., irrigation pond, lake or river). Following this verification step, we focused on landscapes with permanent open-water wetlands comprising less than or equal to 20 percent of the landscape with five equal intervals (>0-4, 4.1-8, 8.1-12, 12.1-16,16.1-20). This new subset (LenW20) resulted in 1198 and 783 landscape samples in the Central and Southern inventories respectively.

During this preliminary investigation, the difference between the land-cover data (AAFC) and the ASRD wetland inventories was explored. We defined disturbance as the amount of developed and agriculture land-cover data in the AAFC dataset and found that 297 samples in the LenW20 subset were recorded as having 100% disturbance and were subsequently removed from the dataset. The removal of these samples resulted in 984 and 700 LenW20 samples in the Central and Southern inventories (Table 2.2).

2.4.2 Landscape metric analysis on final subset

Previous research identified 10 representative landscape metrics for quantifying landscape structure in wetland-rich landscapes (Table 2.3; Evans et al. 2017), which can be organized conceptually into three groups: Shape, Aggregation, and Diversity metrics (McGarigal et al. 2012). Shape metrics characterize configuration at the patch level and can be summarized for all the patches in each land-cover class as area-weighted mean (McGarigal et al. 2012). Aggregation is a measure of the spatial relationship between patches in terms of their isolation, dispersion, interspersion and subdivision (Evans et al. 2017). Diversity metrics are aspatial as they represent the proportions of land-cover classes within a landscape, as opposed to the location and size of patches (McGarigal et al. 2012). The shape, aggregation, and diversity of wetlands metrics were derived from the ASRD wetland data and the diversity of land-cover from the AAFC data. Our final subset of LenW20 landscapes were analyzed with nine of the ten suggested landscape metrics in FRAGSTATS (McGarigal et al. 2002). Patch density of open-water wetlands (PD_OW) was removed from the list due to our preselection for wetland-rich landscapes comprising 20 percent or less permanent open-water wetland. The values of the nine metrics were then analyzed to determine if they differ across inventory and natural region and whether they can be utilized in aiding reclamation activities in areas of ongoing resource extraction. See Appendix 5 for metric formulas and description.

Table 2. 2: Resulting landscape samples for statistical analysis. 13,676 randomly generated landscapes resulted in a final subset of 1,684 LenW landscapes comprising less than or equal to 20% permanent open-water with 984 and 700 in the Central and Southern inventories respectively. LenW20 Represents this final subset with the 100% disturbance landscapes removed.

	Inventory		
	Central	Southern	Total
Landscapes Generated	4597	9079	13676
ASRD Wetlands	4019	5814	9833
LenW	1198	783	1981
LenW Zero Disturbance	227	104	331
LenW with 100% Disturbance*	214	83	1657
LenW20	**984**	**700**	**1684**
Boreal	632	-	-
Parkland	352	141	-
Grassland	-	559	-

* Removed from further analysis

Table 2. 3: 10 parsimonious landscapes representing the spatial configuration of wetlands independent of wetland proportion (Evans et al. 2017). PD_OW was removed from our study due to the pre-selection for wetland-rich landscapes comprising 20% or less permanent open-water.

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 neighbor of wetland patches	ENN_WET
	Patch Density of open-water	PD_OW
	Splitting index for wetlands	SPLIT_WET
Diversity	Simpson's diversity index for wetlands	SIDI_WET
	Simpson's diversity for land cover	SIDI_LAND

2.5 Statistical Tests

To determine if the metric values of the sample landscape differed significantly among disturbance levels, and across inventories and natural regions, results were qualitatively compared using boxplots and their distributions were further analyzed quantitatively. The data were determined to be non-parametric both quantitatively with a Shapiro-Wilk's test (Shapiro & Wilk, 1965) and graphically with Q-Q plots (Wilk & Gnanadesikan, 1986). A Brown-Forsythe test determined that some landscape metrics within the six subsets had unequal variances (Brown & Forsythe, 1974). Where unequal variances were revealed, the Kolmogorov-Smirnov (K-S) test was applied within (one sample test) and between groups (two sample test) manually into a pairwise comparison (Darling, 1957).

The Kruskal-Wallis (K-W) test was applied to all landscape metric comparisons where metric distributions were equal in variance across subsets to determine if metrics were sensitive to changes in disturbance and natural region (Kruskal & Wallis, 1952). In cases where significant differences were found, a Dunn's pairwise comparison was used to determine where those sensitive metrics were occurring among the disturbance intervals (Dunn, 1964). The results of both pairwise tests were combined into a final table to illustrate landscape patterns and a confidence interval (CI), which was calculated with an alpha of 0.05 and confidence level of 95%, is reported for each comparison (See Appendix 6 for CI table).

3. Results

3.1 Distribution of permanent open-water wetlands and disturbance in study area

The Boreal and northern Parkland region of our study area is dominated by peatlands (Downing & Pettapiece, 2006), however, the Central inventory that covers this area demonstrates that there is more permanent open-water wetland-rich landscapes relative to the Southern inventory, where wetlands are mainly open-water depressional wetlands and marshes (Downing & Pettapiece, 2006). For example, there are 984 permanent open-water wetland landscapes identified in the Central inventory and 700 in Southern inventory (Figure 2.3), despite the Southern inventory covering a larger area. This is likely the result of the differing methods used in the creation of the inventories (Central and Southern) as well as increased disturbance (i.e., developed and agriculture lands) in the south.

Among our sample landscapes, 86.1% of samples in the Central inventory and 85.3% of the samples in the Southern inventory had less than 8% of their total area classified as permanent open-water wetlands. Disturbance for most sample landscapes was in the low (0-20%) or high (80-99.9%) disturbance intervals. In the Central inventory, this represented 68.8% of the 984 landscapes samples and in the Southern inventory this represented 64.3% of 700 landscape samples. See Appendix 4 for boxplots of LenW samples with 20 percent or less permanent open-water.

3.2 Analysis of metric values and disturbance in the Central and Southern inventories

The Central and Southern inventories exist along a latitudinal gradient from the Boreal to the Grassland with both inventories including portions of Parkland. We first analyzed our results by inventory to determine if similarities exist across the inventories. The results of a qualitative analysis of the nine metrics for the Central inventory produced limited visual differences in the distributions and median values in the boxplots, except for the diversity metrics, and between the low (0-20%) disturbance landscapes and high (80-99.9%) disturbance landscapes for two of the aggregation metrics: SPLIT and AI (see Appendix 7 for boxplots). Despite limited visual variation, Kruskal-Wallis and Kolmogorov-Smirnov tests illustrated that significant ($p < 0.001$) differences exists across disturbance intervals for six of the nine metrics in the Central inventory (Table 2.4).

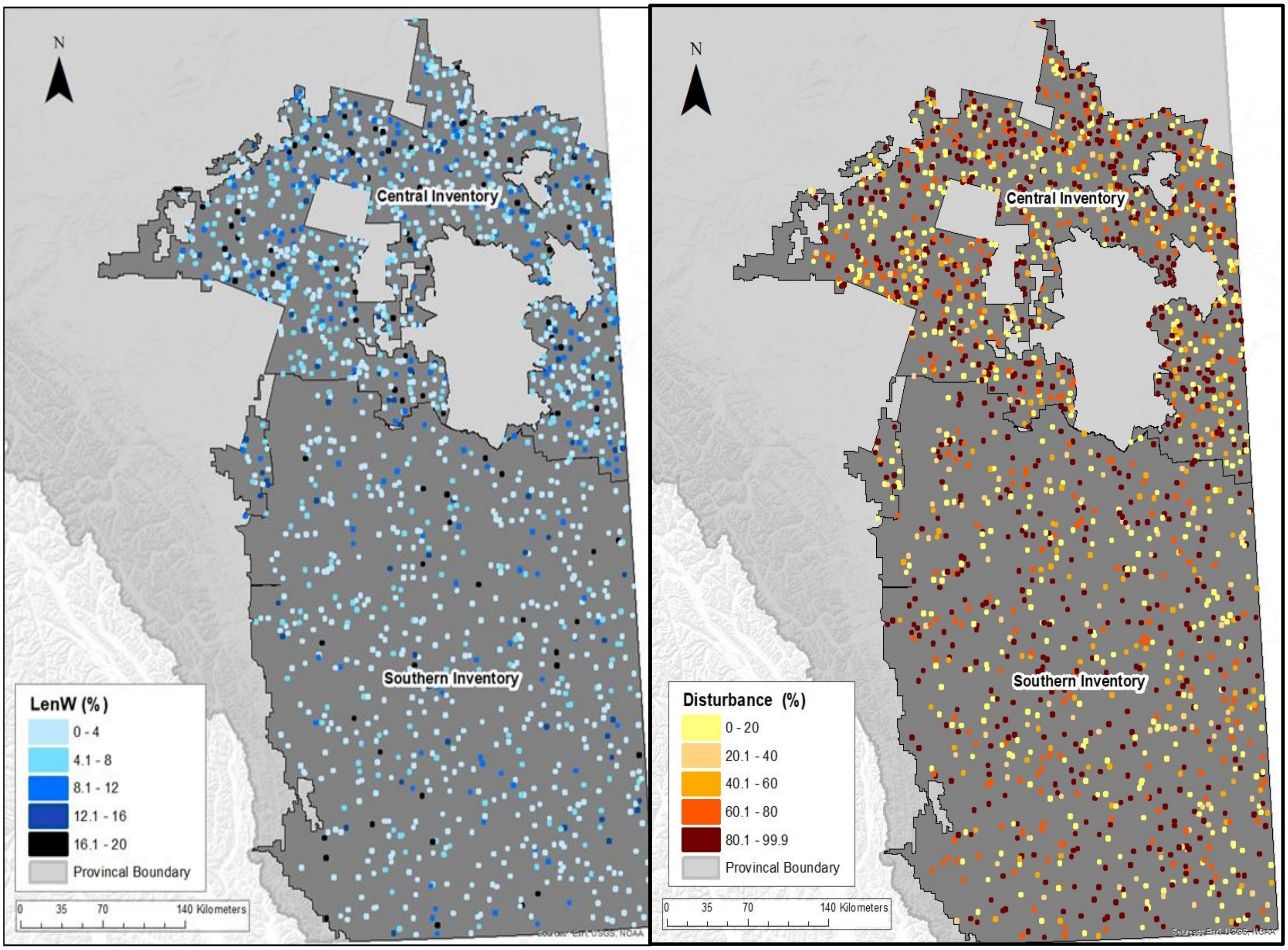


Figure 2. 3: The distribution of permanent open-water (LenW) and disturbance in generated sample landscapes. Sample landscape symbols have been enlarged for increased visibility.

Table 2. 4: Kruskal-Wallis and Kolmogorov–Smirnov test results for the significance of differences between metric values and disturbance in the Central inventory. Six metrics had significant values (p-value <0.001). Four aggregation metrics: AI, COHESION, CONTAG, and SPLIT; two diversity metrics: SIDI_wet and SIDI_land.

Type	Metric	Test Applied	p-value
Shape	CIRCLE_AM	K-W	0.085
	SHAPE_AM	K-W	0.08
Aggregation	AI	K-W	**<0.001**
	COHESION	K-S	**<0.001**
	CONTAG	K-W	**<0.001**
	ENN	K-W	0.563
	SPLIT	K-W	**<0.001**
Diversity	SIDI_wet	K-W	**<0.001**
	SIDI_land	K-S	**<0.001**

To determine if there are significant differences between specific disturbance intervals in the Central inventory, a pairwise comparison was conducted on combinations of disturbance intervals for each metric. Results demonstrated that 40 of 90 comparisons comprising nine metrics compared across five disturbance intervals had significant differences in metric values (Table 2.5). Among these 40 comparisons, 28 were the result of comparisons between the highest disturbed (80-99.9%) landscapes and landscapes with other levels of disturbance. The only metric that had consistently significant metric values and decreased with increasing disturbance levels was diversity of land-cover types (SIDI_land) (p-value <0.001). While low (0-20%) disturbance landscapes did yield 16 metric comparisons that were statistically significant across disturbance levels, seven of these were with the highest level of disturbance. Therefore, there is little difference between landscape structure in low disturbance landscapes and those with disturbance up to 80%, except for land-cover diversity in the Central inventory. AI and COHESION were the most common identified aggregation metric in the low and high disturbed landscapes.

Similar to the Central inventory, distributions and medians for the metrics of the boxplots produced for the Southern inventory did not demonstrate visually obvious differences, except for the diversity metrics and between the low (0-20%) and high (80-99.9%) disturbed landscapes for two aggregation metrics, COHESION and AI (Appendix 7). However, application of the Kruskal-Wallis and Kolmogorov–Smirnov identified eight of the nine metrics as having significantly different metric values among disturbance intervals (p-value < 0.05) (Table 2.6).

Table 2. 5: The results from the Dunn's and Kolmogorov–Smirnov pairwise comparison between disturbance intervals for the Central inventory. Statistical power was calculated with an alpha of 0.05 (confidence level of 95%). The confidence interval (CI) is reported under sample size (n) for each disturbance interval.

Disturbance class (%)	**0-20** n=295 CI=5.71	**20-40** n=56 CI=13.1	**40-60** n=102 CI=9.7	**60-80** n=149 CI= 8.03
20-40 n=56 CI=13.1	CIRCLE_AM* SIDI_land***			
40-60 n=102 CI=9.7	COHESION* SIDI_land*** AI*	SIDI_land*		
60-80 n=149 CI=8.03	COHESION** SPLIT** AI SIDI_land***	SIDI_land***	SIDI_land***	
80-99.99 n=382 CI=5.01	SHAPE_AM* COHESION*** SIDI_land*** SIDI_wet*** CONTAG*** SPLIT*** AI***	SHAPE_AM* CIRCLE_AM* COHESION** SIDI_land*** SIDI_wet** CONTAG*** SPLIT*** AI**	SHAPE_AM* COHESION* SIDI_land*** SIDI_wet*** CONTAG** SPLIT*** AI**	COHESION** SIDI_land*** SIDI_wet* CONTAG* SPLIT*** AI***

* p=<0.05; **p=<0.01; ***p<0.001

Table 2. 6: Kruskal-Wallis and Kolmogorov–Smirnov test results for the significance of differences between metric values by disturbance intervals in the Southern inventory. Results show that eight of the nine metrics vary with disturbance.

Type	Metric	Test Applied	p-value
Shape	CIRCLE_AM	K-S	**<0.001**
	SHAPE_AM	K-S	**<0.001**
Aggregation	AI	K-S	**<0.001**
	COHESION	K-S	**<0.001**
	CONTAG	K-W	**0.027**
	ENN	K-W	0.94
	SPLIT	K-S	**<0.001**
Diversity	SIDI_wet	K-S	**<0.001**
	SIDI_land	K-S	**<0.001**

A pairwise comparison was used to compare landscape structure between disturbance intervals as was done for the Central inventory. Results for the Southern inventory were similar but less systematic than that of the Central inventory and demonstrate that 26 of 90 comparisons of nine metrics across five disturbance intervals had significantly different metric values. Of the 26 comparisons yielding significant differences, 19 were the results of comparisons between the highest (80-99.9%) disturbed landscapes and landscapes with other levels of disturbance. Again, land-cover diversity was the only metric that had consistently significant different metric values across all disturbance levels.

Overall, analysis of landscape patterns in the Central and Southern inventories displayed similar results, whereby landscapes in the high (80-99.9%) disturbance interval were significantly different from other landscapes and identified the same six metrics. Four aggregation metrics: AI, COHESION, CONTAG, and SPLIT, and two diversity metrics: SIDI_wet, and SIDI_land. Metric values for diversity of land-cover were significant in every comparison of disturbance level for both inventories. AI, CONTAG and SPLIT increased with disturbance, while COHESION and SIDI_land decreased with increasing disturbance levels.

3.3 Analysis of metric values and disturbance for the natural regions

To determine if differences in landscape structure exist between natural regions, a comparison of metric distributions between natural regions was conducted. Results were similar to the analysis of metric values and disturbance within the inventories (Central and Southern) in that they displayed limited visual differences in variation and distributions captured in boxplots, nevertheless some visible variation between natural regions was obvious, particularly in the low (0-20%) and high (80-99.9%) disturbance intervals (Appendix 8).

Table 2. 7: The results from the Dunn's test and Kolmogorov–Smirnov pairwise comparison of disturbance intervals for the Southern inventory. Statistical power was calculated with an alpha of 0.05 (confidence level of 95%). The confidence interval (CI) is reported under sample size (n) for each disturbance interval.

Disturbance class (%)	**0-20** n=132 CI =8.53	**20-40** n=64 CI=12.25	**40-60** n=69 CI=11.8	**60-80** n=117 CI=9.06
20-40 n=64 CI=12.25	SIDI_land*** SPLIT*			
40-60 n=69 CI=11.8	SIDI_land***	SIDI_land**		
60-80 n=117 CI=9.06	SIDI_land***	SIDI_land**	SIDI_land***	
80-99.99 n=318 CI=5.5	CIRCLE* COHESION*** SIDI_land*** SIDI_wet** CONTAG* SPLIT***	SIDI_land***	CIRCLE* SIDI_land*** SIDI_wet** SPLIT** COHESION**	AI** CIRCLE_AM*** COHESION** SIDI_land*** SIDI_wet** SPLIT*** AI**

* p=<0.05; **p=<0.01; ***p<0.001

Kruskal-Wallis and Kolmogorov-Smirnov tests were used to compare the metric values between natural regions (Boreal and Parkland) in the Central inventory of equivalent disturbance intervals. Results produced just three significantly different metric values of 45 comparisons: SHAPE_AM, COHESION, and SPLIT (p-values 0.016, <0.001, <0.001) respectively (Appendix 9). The same analysis in the Southern inventory between equivalent disturbance intervals for the

Grassland and Parkland regions produce three significantly different metric values for CIRCLE_AM, SPLIT, and SIDI_wet (p-values 0.013, <0.001, <0.001) respectively, all three were in the 40-60% disturbance interval (Appendix 9). These limited results were likely due to the lack of data for the central Parkland and the shared characteristics between the southern Boreal and north-west Parkland and likewise, between the southern Parkland and Grassland region in the southern inventory.

While the landscapes between natural region showed little significant difference within the same disturbance class, we sought to determine if landscapes differed with disturbance within a specific natural region. Results from the application of the Kruskal-Wallis and Kolmogorov-Smirnov tests to the Boreal region yielded six of nine metrics demonstrating significantly different results across disturbance levels. For the Parkland region (Central inventory), significant differences among landscapes comprising different levels of disturbance were also observed, but these were captured by only four of nine metrics (Table 2.8).

In the Southern inventory, the Kruskal-Wallis and Kolmogorov-Smirnov tests were used to compare across disturbance levels for the Parkland and Grassland regions. Within the Parkland, only one metric value (SIDI_land) significantly differed among disturbance intervals, while eight metric values (SHAPE_AM, CIRCLE_AM, AI, COHESION, CONTAG, SPLIT, SIDI_wet and SIDI_land) were identified as being significantly different in the Grassland region (Table 2.8).

To further investigate differences across disturbance intervals, pairwise comparisons were made between all combinations of disturbance intervals within each natural region. Results for the Boreal region were similar to those obtained for the Central inventory as a whole, whereby landscapes comprising a high level of disturbance (80-99.9%) were significantly different from other landscapes. Little difference was found between landscapes with 0-80% disturbance with the exception of land-cover diversity (SIDI_land) (Table 2.9).

The same analysis applied to the Parkland region (Central inventory) displayed a less systemic pattern of differences among landscapes by disturbance interval. However, sample landscapes in the high disturbance interval (80-99%) still maintained the most difference from other landscapes. Metric values for diversity of the land-cover (SIDI_land) had a significant (p-value <0.01) value in every comparison (Table 2.10).

Table 2. 8: Kruskal-Wallis and Kolmogorov -Smirnov tests were conducted on the natural regions (Boreal, Parkland, Grassland) separately within each inventory (Central and Southern).

	Central Boreal		Central Parkland		Southern Parkland		Southern Grassland	
Metric	**Test Applied**	**p-value**	**Test Applied**	**p-value**	**Test Applied**	**p-value**	**Test Applied**	**p-value**
CIRCLE_AM	K-W	0.411	K-W	0.233	K-S	0.84	K-S	**<0.001**
SHAPE_AM	K-W	0.089	K-W	0.168	K-S	0.462	K-S	**<0.001**
AI	K-W	**<0.001**	K-W	**0.002**	K-S	0.831	K-S	**<0.001**
COHESION	K-S	**<0.001**	K-S	**<0.001**	K-S	0.613	K-S	**<0.001**
CONTAG	K-W	**<0.001**	K-W	0.416	K-W	0.802	K-W	**0.005**
ENN_MN	K-W	0.759	K-W	0.848	K-W	0.872	K-W	0.741
SPLIT	K-S	**<0.001**	K-W	**<0.001**	K-S	0.616	K-S	**<0.001**
SIDI_wet	K-W	**<0.001**	K-W	0.395	K-S	0.774	K-S	**<0.001**
SIDI_land	K-S	**<0.001**	K-S	**<0.001**	K-S	**<0.001**	K-S	**<0.001**

Table 2. 9: The results the pairwise comparison of disturbance intervals for the Boreal region of the Central inventory. Statistical power was calculated with an alpha of 0.05 (confidence level of 95%). The confidence interval (CI) is reported under sample size (n) for each disturbance interval.

Disturbance class (%)	**0-20** n=188 CI=7.15	**20-40** n=38 CI=15.9	**40-60** n=59 CI=12.76	**60-80** n=98 CI=9.9
20-40 n=38 CI=15.9	CIRCLE_AM* SIDI_land***			
40-60 n=59 CI=12.76	SIDI_land*** AI*	SIDI_land**		
60-80 n=98 CI=9.9	SIDI_land***	SIDI_land**	SIDI_land**	
80-99.99 n=249 CI=6.21	SHAPE_AM* COHESION*** SIDI_land*** SIDI_wet*** CONTAG*** SPLIT** AI***	SHAPE_AM* AI** SIDI_land** SIDI_wet*** CONTAG**	SIDI_land** SIDI_wet*** CONTAG** SPLIT*** AI* COHESION*	SHAPE_AM* COHESION*** SIDI_land** SIDI_wet*** CONTAG** SPLIT*** AI***

* p=<0.05; **p=<0.01; ***p<0.001

Table 2. 10: The results from the pairwise comparison of disturbance intervals for the Parkland region of the Central inventory. Statistical power was calculated with an alpha of 0.05 and a confidence interval of 95%. The confidence interval (CI) reported under sample size (n) for each disturbance interval

Disturbance class (%)	**0-20** n=107 CI=9.47	**20-40** n=18 CI=23.1	**40-60** n=43 CI=14.49	**60-80** n=51 CI=13.72
20-40 n=18 CI=23.1	SIDI_land** SPLIT*			
40-60 n=43 CI=14.49	SIDI_land***	SIDI_land*		
60-80 n=51 CI=13.72	SHAPE_AM* COHESION** SPLIT*** SIDI_land***	SIDI_land***	SHAPE_AM* SIDI_land*** SPLIT**	
80-99.99 n=133 CI=8.5	COHESION*** SIDI_land*** SIDI_wet* SPLIT*** AI***	CIRCLE_AM* SIDI_land***	COHESION* SIDI_land*** SPLIT** AI*	SIDI_land*** AI* COHESION*

* p=<0.05; **p=<0.01; ***p<0.001

When a pairwise comparison of sample landscapes was conducted for the Parkland (Southern inventory), the only significant metric values were diversity of land-cover (SIDI_land), which had significant (p-value <0.001) values in all disturbance intervals and was the only metric identified consistently in both Parkland comparisons (i.e., Central and Southern inventory Parkland; Appendix 10). In contrast, landscape patterns in the Southern Grassland were affected by disturbance levels. However, low (0-20%) disturbance landscapes significantly differed from landscapes in other disturbance intervals with 19 metrics having significant values and high (80-99.9%) disturbance landscapes retaining 17 significant metrics. Land-cover diversity was significantly different among nearly all disturbance intervals except the 20-40% interval where the limited samples and statistical power likely played a role in detecting significant metric values in this subset (Table 2.11).

Table 2. 11: The results of the pairwise comparison on disturbance intervals for the Grassland region of the Southern inventory. Statistical power was calculated with an alpha of 0.05 and a confidence interval of 95%. The confidence interval (CI) reported under sample size (n) for each disturbance interval

Disturbance class (%)	**0-20** n=107 CI=9.47	**20-40** n=18 CI=23.1	**40-60** n=43 CI=14.95	**60-80** n=51 CI=13.72
20-40 n=18 CI=23.1	AI*** COHESION*** SPLIT** SIDI_land*** SIDI_wet***			
40-60 n=43 CI=14.95	AI*** COHESION*** SIDI_wet*** SIDI_land***	NA		
60-80 n=51 CI=13.72	AI*** COHESION*** SIDI_wet SIDI_land***	NA	SHAPE_AM* SIDI_land*** SPLIT**	
80-99.99 n=133 CI=8.5	COHESION*** SIDI_land*** SIDI_wet*** CIRCLE_AM** SPLIT** AI***	NA	CIRCLE_AM*** SIDI_land*** SPLIT** SIDI_wet**	CIRCLE_AM*** SHAPE_AM** AI* COHESION** SPLIT*** SIDI_wet*** SIDI_Land***

* p=<0.05; **p=<0.01; ***p<0.001

3.4 Association of average metric values and landscape structure

In seeking to understand the landscape structure associated with permanent open-water wetlands within the natural regions, we compared the average metric values for all nine metrics across both inventories (Central and Southern) and across their respective natural regions (Boreal, Parkland, Grassland). Average metric values showed varying responses across inventories and latitudinally across the natural regions, but generally the complexity and diversity of permanent open-water wetland-rich landscapes decreased across a north-to-south gradient.

Shape metrics provide insight about the complexity of permanent open-water wetlands. For example, SHAPE_AM is a measure of overall shape complexity (McGarigal & Marks, 1995) and when the value of SHAPE is equal to one, the wetland patch is maximally compact (square or nearly square). The value of SHAPE rises without limit as the shape becomes more irregular (McGarigal & Marks, 1995). The average shape metric values decreasing or becoming closer to one in the south indicate that the wetlands are becoming more compact and less complex (Table 12). CIRCLE_AM is the smallest circumscribing circle within in the wetland patch and is a measure of overall patch elongation. Values for CIRCLE_AM are between zero and one, with values closer to one representing patches that are more elongated (McGarigal & Marks, 1995). The average values for CIRCLE_AM decreasing in the south further confirms that permanent open-water wetlands are becoming less complex or elongated in the south, where there is generally more disturbance and consolidation and drainage of wetlands in agriculture (McCauley, 2015; Table 2.12).

The average values of aggregation metrics across inventories and natural regions supports the findings that permanent open-water wetlands are more compact and less complex along a north-to-south gradient. For example, ENN_MN uses Euclidean distance as a measure of patch isolation (McGarigal & Marks, 1995). In the south, these values are smaller, indicating permanent open-water wetland patches are closer together. CONTAGION is widely used in landscape ecology as a measure of clumpiness on categorical maps (Tuner, 1989) and incorporates both dispersion and interspersion in its calculation (McGarigal & Marks, 1995). The general increase in values along the north-to-south gradient indicates that permanent open-water wetlands are more clumped together and occupy a slightly larger percent of the sample landscapes. COHESION represents the physically connectedness of patches (McGarigal & Marks, 1995) and demonstrates that patches of permanent open-water are aggregated or more physically connected in the south. AI is a measure of patch adjacency (McGarigal & Marks, 1995). Adjacency to other permanent open-water wetland patches increases in the south, again confirming that permanent open-water wetlands are becoming more compact, and less elongated and complex. SPLIT is measure of fragmentation and can be interpreted as the effective mesh number, where when SPLIT is equal to one, it consists of a single patch (Jaeger, 1999). The values for this metric decrease substantially along a north-to-south gradient, further supporting the general trend of permanent open-water wetlands being more compact and aggregated in the south.

Metric values representing the diversity of permanent open-water wetland patches (SIDI_wet) decrease from Boreal to Grassland, which indicates that composition of those wetland patches increasingly trends towards a single patch (McGarigal & Marks, 1995). Finally, the change in average values for diversity of land-cover (SIDI_land) signifies that landscape composition surrounding permanent open-water wetlands also aggregates towards larger patches, indicative of the presence of agriculture in the south.

The general patterns observed suggest that permanent open-water wetlands become increasingly compact, aggregated, and less complex and elongated along a north-to-south gradient; however, differences in the methodologies used to create inventories is also a factor. Figure 2.4 provides examples of various landscape patterns and their associated characteristics and metric values.

Table 2. 12: Average metric values for the inventories (Central and Southern) and their respective natural regions (Boreal, Parkland, Grassland).

		Metrics								
		Shape		Aggregation					Diversity	
Inventory	**Natural Region**	**SHAPE_ AM**	**CIRCLE_ AM**	**ENN_ MN**	**CONTAG**	**COHESION**	**AI**	**SPLIT**	**SIDI_ wet**	**SIDI_ land**
Central		*2.385*	*0.694*	*186.016*	*63.183*	*96.896*	*91.192*	*3124.944*	*0.517*	*0.367*
	Boreal	2.400	0.693	186.379	62.984	96.880	91.105	967.348	0.520	0.368
	Parkland	2.357	0.696	185.366	63.540	96.925	91.350	6998.810	0.513	0.366
Southern		*1.765*	*0.443*	*127.966*	*77.589*	*99.629*	*98.717*	*1.335*	*0.144*	*0.325*
	Parkland	1.703	0.417	120.141	88.408	99.687	98.877	1.179	0.089	0.303
	Grassland	1.770	0.445	128.586	76.731	99.625	98.705	1.348	0.148	0.327

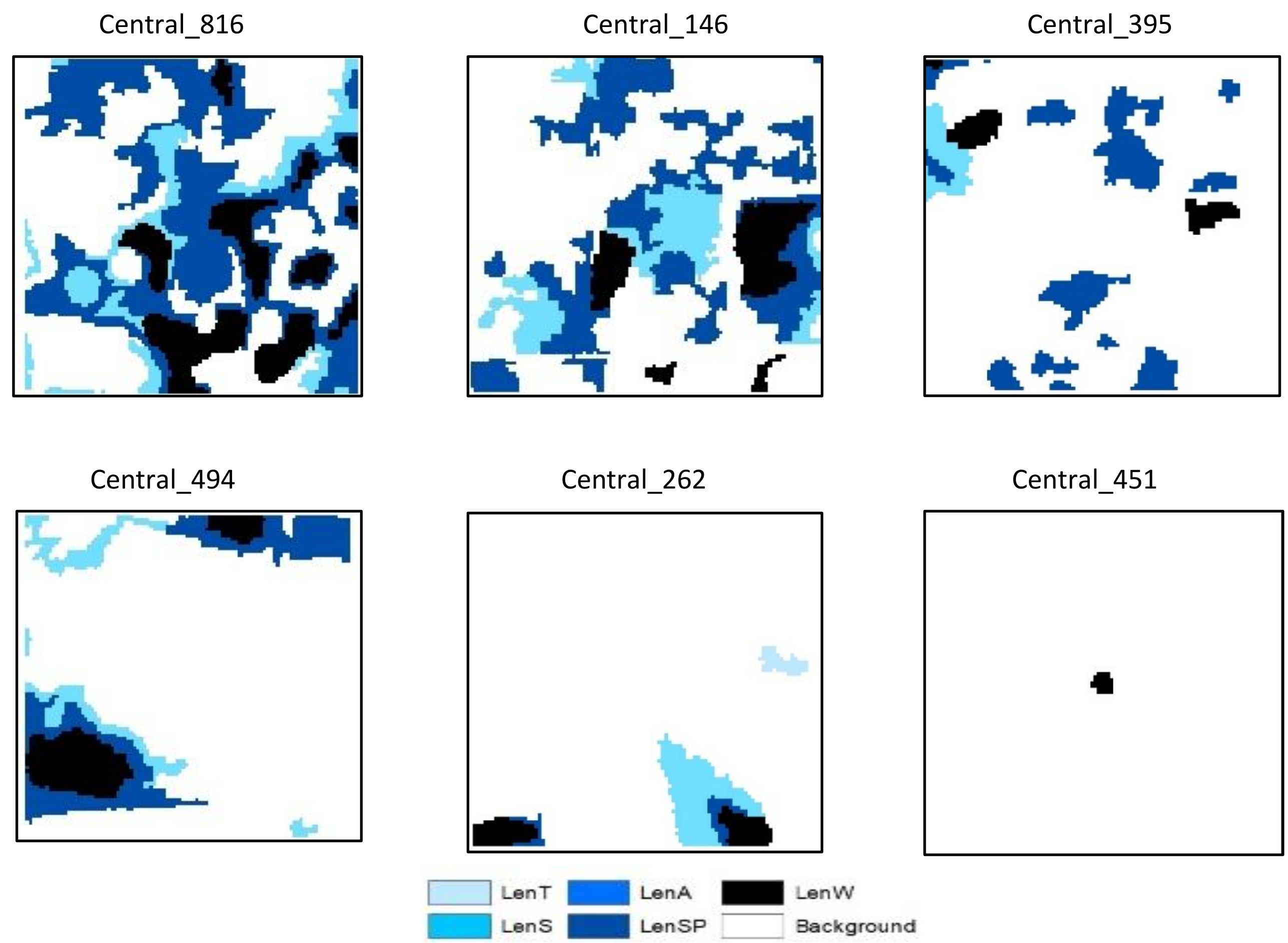


Figure 2. 4: Six examples of landscape patterns from the Central inventory. The top three show increasing disturbance from left to right while the bottom three show the effects of different dominate land-cover

Table 2. 13: Characteristics and metric values for the six sample landscapes from Figure 4.

ID	Central_816	Central_146	Central_395	Central_494	Central_2262	Central_4518
Perc. LenW	13. 31	6.96	2.34	6.42	2.23	0.39
Perc. Disturb	0	70.43	98.71	90.63	97.88	0
Land-cover	Shrub 33.15, Forest 54.36,	Ag, 70.43, Shrub 4.49,	Ag 98.71	Ag, 90.63	Ag 11.29 Developed 86.59	Forest 83.93
CIRCLE_AM	0.731	0.672	0.589	0.73	0.62	0.329
SHAPE_AM	3.988	2.605	1.662	2.465	1.732	1.077
AI	86.334	88.333	88.839	90.151	92.064	98.462
COHESION	97.451	96.199	93.543	96.809	95.396	100
CONTAG	50.53	55.223	63.964	50.145	54.372	100
ENN_MN	71.007	133.846	121.936	314.902	488.648	0
SPLIT	18.841	64.045	394.21	102.198	552.135	65746.22
SIDI_wet	0.573	0.572	0.464	0.633	0.635	0
SIDI_land	0.579	0.212	0.025	0.17	0.237	0.27

4. Discussion

The concept of managing ecosystems in a manner consistent with their undisturbed structure and process was widely applied by landscape managers and government agencies for ecological monitoring in the 1990s (Christensen et al. 1996; Bowman & Somers, 2005; Pardo et al. 2012). During this time, phrases such as, ecosystem integrity, resiliency, and biodiversity became synonymous with the goal of ecosystem management — to create healthy and sustainable ecosystems that can preserve their structure and organization over time (Grumbine, 1994; Whitford & deSoyza, 1999). The quantification of abiotic and biotic elements in low disturbance landscapes allows for the use of a benchmark or reference condition approach to evaluate and compare the health of ecosystems (Hawkins et al. 2010).

Evaluating landscapes with a reference condition approach typically involves the comparison of biotic indices. A similar method called Historical Range and Variability (HVR) has been used to evaluate landscape pattern, primarily in forested landscapes, with historical evidence obtained from remotely sensed imagery (e.g., air photos, satellite photos) (Albella & Denton, 2009; Keane et al. 2002). Applications of HRV in wetland-rich landscapes are limited, however, the work of Liu and Cameron (2001) and Li et al. (2010) observed changes in the complexity, shape, and aggregation of wetlands with increasing anthropogenic disturbance. While our results and those of Evans et al. (2017) confirm these conclusions, the practicality of using such methods in wetland-rich landscapes is debatable as a major challenge with any HRV or a reference condition approach is selecting the appropriate historical reference date and condition (e.g., level of natural variability and disturbance) (Stoddard et al. 2006; Keane et al. 2009). Furthermore, in many cases, using a reference condition or HRV approach to design and evaluate reclaimed wetland landscapes may be unrealistic as it may not be possible to return a landscape back to its original state (e.g., Johnston et al. 2016).

To provide insight to reclamation planning and closure permitting, this exploratory analysis presented a methodology to quantify permanent open-water wetland landscape structure across multiple levels of disturbance and across three natural regions. We then conducted several statistical comparisons to determine if landscapes differed between disturbance levels and across natural regions and inventories. However, due to the nature of our data (i.e., non-parametric, unequal sample sizes and variances), we were forced to use separate methods for determining statistical significance due to unequal variance (i.e., Kruskal-Wallis and Kolmogorov-Smirnov), which may have lowered our statistical power, particularly in the Southern inventory where sample sizes were small. As a result, consistent and distinct landscape pattern changes in permanent open-water wetlands across disturbance levels, and across inventories and natural regions were limited.

Reclamation of wetland-rich landscapes associated with mega-projects tend to substantially alter local topographic conditions. The costs of recreating initial topographic conditions are often prohibitive and costly (Lemphers et al. 2010), thus if natural landscapes can maintain a specific extent of permanent open-water wetlands, then modeling reclaimed wetlands based on natural permanent open-water should enable us to build reclaimed landscapes where the surrounding landscape can maintain the wetlands. While landscapes in Alberta do comprise permanent open-water wetlands, of the 1,684 open-water landscapes in this study, the area classified as permanent open-water in our sample landscapes was predominately low with the majority (86.1% and 85.3% in the Central and Southern inventory respectively) having 8% (0.08 km^2) or less of their area in permanent open-water (LenW) wetlands.

Our quantitative comparison of permanent open-water wetland-rich landscapes across disturbance levels and natural regions identified distinct landscape patterns. Among the disturbance levels, we found significant differences between landscapes comprising 0-80% and 80-99.9% in the Boreal and Parkland and among the low (0-20%) landscapes in the Grassland region. While statistically significant differences in landscape pattern were present among the disturbance intervals between 0-80% in the Boreal and Parkland, these were less systematic. When comparisons were made across natural regions, holding disturbance level constant, landscape patterns were not significantly different. However, we did not make comparisons directly between Boreal and Grassland regions due to differences in data resolution and quality. These results indicate the existence of four distinct landscape patterns: 1) Boreal 0-80% disturbed, 2) Boreal 80-99.9% disturbed 3) Grassland 0-20% disturbed, and 5) Grassland 80-99.9% disturbed.

The lack of identifiable landscape patterns in the low (0-20%) disturbed landscapes in the Boreal and Parkland was interesting given that 30% of samples in the Central inventory and 18.9% of samples in the Southern inventory were in this lower disturbance interval. This is likely a result of the unique relationship permanent open-water has to the surrounding topography in this study area and further investigation of the sensitivity of metric values to data quality is required.

Issues with data quality between the Central and Southern inventory were highlighted by contrasting results for the Parkland region. Higher accuracy levels and statistical power associated with the Central inventory data lead to significantly different levels of wetland aggregation across disturbance intervals, which was not observed for the Southern inventory While differences among the disturbance levels exist for the Parkland region, results for the Parkland from both inventories were less systematic than those found for the Boreal and Grassland regions. The lack of significant differences in Parkland versus Boreal or Grassland regions is likely due to its position between the two, whereby it acts like an ecotone sharing common landscape characteristics with both the Grassland and Boreal natural regions as well as being highly disturbed by agriculture (Downing & Pettapiece, 2006). Further, the central portion of the Parkland region was not captured by either inventory. This gap in the central portion of the Parkland, and the lower statistical power of the Southern inventory is a likely factor for the lack of differences between the Parkland and other regions.

The exception to a lack of differentiation between the Parkland and Boreal or Grassland regions as well as differences across levels of disturbance is the diversity of land-cover (SIDI_land), which was statistically significant in every disturbance comparison across inventories and natural regions. The statistical significance of the metric values for SIDI_land across all disturbance intervals and natural regions may be partly explained by the small portion of permanent open-water wetlands occupying the sample landscapes, while other land-cover types account for a larger proportion of the samples. Additionally, there is substantial variation in vegetation across the natural regions and the dominate disturbances within. For example, in the Grassland region, there is an average of 4.5% wetland land-cover that is surrounded by natural grasses and primarily disturbed by urbanization, oil and gas, and irrigation farming. The Parkland region has slightly more wetlands on average (6.7%) that are surrounded by limited native landscape, which is mostly of shrubland, and primarily disturbed by urbanization, till cropping, and oil and gas, whereas the Boreal region has significantly more wetlands on average (24.8%) that are surrounded by Boreal forest, and mainly disturbed by oil and gas, forestry, coal mining, and grazing (Appendix 1; Wray & Bayley, 2006; Downing & Pettapiece, 2006).

Our results demonstrate that the metrics produced by Evans et al. (2017) were able to differentiate a subset of landscapes found in Alberta (i.e., permanent open-water wetland-rich landscapes), whereby eight out of the nine applied landscapes metrics had values that were statistically different across disturbance intervals in several comparisons. The spatial distribution of permanent open-water wetlands (ENN_AM) was not significant in any comparison. In the study by Evans et al. (2017) all wetland types were considered (i.e., permanent open-water, semi-permanent, temporary and seasonal), whereas in our study only permanent open-water wetlands were analyzed. Average values for ENN_AM in our analysis were 186.02 m and 127.96 m in the Central and Southern inventories respectively, whereas in the study by Evans et al. (2017) values were smaller at 108.52 m and 135.91 m in the Central and Southern inventories respectively. ENN_MN was found to be statistically different among the disturbance intervals in several subsets in the study by Evans et al. (2017), which can be attributed to the fact there are more semi-permanent and temporary wetlands in our study area, which are more susceptible to disturbance (Serran & Creed, 2014). This knowledge combined with the size, permanence, dispersion and fact many of these wetlands are considered geographically (Mushet et al. 2015), it is likely that the distance between patches of permanent open-water wetlands are less susceptible to disturbance and their distribution is more a result of the extensive history of glaciation in this region and the resulting effects on topography (Johnson et al. 2005).

Landscape ecologists understand spatial heterogeneity is scale dependent and the extent at which we conduct landscape analysis effects the metrics used to quantify patterns (Turner, 1989; Wu, 2004). Therefore, it should be noted that while the 1 km^2 landscape sample size accurately captured the disturbance footprint in Alberta, this may not be true for other regions involved in landscape-level reclamation and changing the scale and extent of the analysis with ultimately affect the metrics, patterns, and processes identified in landscape.

4.1 Climate and wetland-rich landscape reclamation

Hydrology is often attributed as the most important factor influencing wetland development, chemistry and ecology (Winter, 1989 & 1992; Mitsch & Gosselink, 1993; Sanchez-Carrillo et al. 2004; Hayashi et al. 2016). While the foundation and location of hydrology is a function of topography, the magnitude and frequency of hydrological flows is a function of climate (Euliss et al. 1999; Erwin, 2009). In the semi-arid regions of Alberta, within which our study lies, potential evaporation exceeds precipitation and the water balance of wetlands in this region is sensitive to the effects of changing climate and land-use practices in their surrounding uplands (Conly & van der Kamp, 2001). Further, the hydrology of these wetlands are important focal points for groundwater recharge due to the snow melt they retain from both an ecological (van der Kamp & Hayashi, 1998; Conly & van der Kamp, 2001) and economic perspective (i.e., importance to agriculture, ecosystem services; Ameli & Creed, 2018).

Canadian climate models and the studies conducted using their various climate scenarios predict increases in extreme temperatures and precipitation (Tebaldi et al. 2006; Pike et al. 2008; Kompanizare 2018). Using the mean changes in precipitation and climate, further studies forecast important changes to the hydrological cycle: an earlier snowmelt, decrease in snowpack depth and length of the snow season, increase in stream flow during the start of the growing season, increase in the length of growing season (Pike et al. 2008; Zhang et al. 2001; Kompanizare et al. 2018), and increase in evapotranspiration (Pan et al. 2015). This has important implications for wetlands and the role permanent open-water wetlands will play in mega-project reclamation, particularly with respect to the hydrology of reclaimed permanent open-water wetland-rich landscapes in mining areas where a thinner surficial geological layer makes them infrequently connected and more susceptible to disturbance from mining activities (Kompanizare et al. 2018).

Given the lengthy timelines of mega-projects like the oil sands, reclamation in wetland-rich landscapes must consider future climate to produce landscapes that are self-sustaining and resemble their natural counterparts. Global Climate Models (GCMs) predict that Grassland and Parkland climates will shift northward and replace much of the Boreal (Lemieux & Scott, 2005; Schneider, 2013; Schneider et al. 2015). Therefore, instead of using historical or low disturbance landscapes in the Boreal as guidelines or benchmarks for closure planning and approvals, it has been proposed that landscapes currently within the projected climate envelopes are more appropriate targets for wetland-rich landscape reclamation (Rooney et al. 2015). Consequently, understanding the structure of permanent open-water landscapes in the Grassland and Parkland natural regions is likely more informative for permanent open-water wetland reclamation in the southern Boreal than a traditional HRV or reference condition approach.

In addition to direct interaction between open-water and climate, changing proportions of wetland types (i.e., from native fens to permanent open-water wetlands) in combination with a changing climate will have considerable ecological implications for the Boreal region. Permanent open-water wetlands with longer, more consistent hydroperiods, typically have less macroinvertebrates, less biodiversity, and more predatory fish than temporary and seasonal wetlands, which are preferred by breeding waterfowl (Mallory et al. 1994). Further, the higher the level of wetland aggregation the greater the surface connectivity between wetlands, which influences hydrological functions and processes such as storage during peak flow (Leibowitz, 2003; Cohen et al., 2016), whereas wetlands that are less permanent and connected are known for having better retention properties because there is less surface water flow to remove pollutants from wetlands (Marton et al. 2015).

We focused on permanent open-water wetland-rich landscapes because earth movement in resource extraction is costly and therefore mega-project plans often retain large open-water wetlands that did not previously exist at the location of resource extraction. We have shown that these landscapes are not only rare but occupy a small proportion of mega-project scale reclamation landscapes. Furthermore, the influence of these novel open-water wetland ecosystems on hydrological connectivity in the face of increasing temperatures and irregular and increased precipitation events is only now being investigated (e.g., Kompanizare et al. 2018). Increased presence of open water in a semi-arid region is likely to lead to increased loss of critical water resources through evaporation, which in similar locations (e.g., south western United States) has been shown to alter local weather patterns (Penman 1948; Lawrimore & Peterson, 2000; Smith et al. 2002).

4.3 Considerations for wetland classification systems

Wetlands are the interface of terrestrial and aquatic systems, which makes them inherently complex and variable (Marton et al. 2015). Furthermore, the functioning of wetlands is influenced by local climate and the resulting variation in hydrology, chemistry, and vegetation, which presents additional challenges to mapping, monitoring, managing, and understanding these important ecosystems (Cohen et al. 2016). These complexities present a difficult classification challenge that has resulted in the creation of a multitude of wetland types that range from permanently flooded open-water wetlands, to temporary ponds, and costal mangroves (Keddy, 2010). While many classification systems exist at the international, regional and local levels across the globe (e.g., RAMSAR (1971), Steward and Kantrud (1971), Cowardin System (1979), Brinson (1993), few include specific recognition of man-made wetlands.

The RAMSAR International Wetland Classification offers an example of one classification system that includes man-made wetlands (Matthew, 1993). However, it is the least applicable because it is a generalization for global application that often translates poorly to local environments and regions (Matthew, 1993). The importance of an appropriate classification system is that it offers simultaneously an improved understanding of the wetland being discussed but perhaps more importantly improved communication about that understanding. Given the extensive number of wetlands expected to be reclaimed as part of mega-projects in Alberta and globally, the need to create a distinct category man-made wetland in wetland classification systems is imperative.

An appropriate classification system will not only help our current understanding, but it will also provide insight about legacy effects and wetland characteristics. For example, while Alberta has policy offering guidance on reclamation to mimic natural topography (CEMA 2014) these guidelines are likely creating similarly shaped wetlands due to recommended height-to-length ratios for slopes (Green Plan Ltd, 2014) and length-to-width ratios for their shape (CH2MHILL 2014). If man-made wetlands existed as part of a classification system, they may be excluded or have their importance reduced so as not to bias scientific conclusions or prescriptive actions and policy. Without classification of these man-made ecosystems, our ability to evaluate, monitor, and compare the functioning and sustainability of these novel ecosystems is impeded. Therefore, a gap remains and there is a need for the identification of man-made wetlands to be included in future wetland classification systems at the local and regional level, to ensure the effective evaluating, monitoring, and comparison of wetlands of any type.

While proposals for improved wetland classification are not new (e.g., Brinson, 1993), it is imperative that consistent and accurate wetland classification data exist. The Canadian Wetland Classification System provides national coverage for Canada; however, the accuracy of this system is low (Finlayson & van der Valk, 1995; Zoltai & Vitt, 1995) and the classification lacks representation of hydroperiod (Zoltai & Vitt, 1995). The two wetland inventories (i.e., Central and Southern) utilized by the presented research augmented these data to derive hydroperiod classification, however, they varied in terms of their accuracy (83%, 51-68%) and classification methods (i.e., predictive ecosystem decision-tree, support vector machine). Given the differences in these data we could not make direct comparisons between landscapes in the Boreal and Grassland regions. Furthermore, a third dataset was required to represent the land-cover surrounding wetlands (AAFC, 2009). A harmonized or single dataset comprising these characteristics would enhance our ability to analyze wetland-rich landscapes.

Despite the variation in our results by wetland inventory for the same natural region (i.e., Parkland), our metric values offer an exploratory guideline for regulators, and academics seeking to design and approve closure plans at the landscape level. However, given these differences, further research is required to determine how robust the presented results are to variation in the spatial resolution and accuracy of wetland and land-cover classification.

5. Conclusion

The quantification of landscape structure is often utilized to understand the interactions between ecological processes and landscape patterns (Turner et al .1989; Wei et al. 2017). The aggregation and connectivity of permanent open-water wetland patches to the surrounding landscape has ecological (Galatowitsch & van der Valk, 1996) and hydrological implications (Cohen et al., 2016) that require their consideration for reclamation projects. Important ecological processes such as, the propagation of seeds, energy and nutrient exchanges (Galatowitsch & van der Valk, 1996), and the presence or absence of avian species, invertebrates, and fish are all affected by the aggregation of wetland patches within the landscape (Haig et al., 1998; Fairbairn & Dinsmore, 2000; Stephens et al., 2005).

Wetlands are valuable, multi-functional ecosystems that can provide an array of subsequent services to humans (see Zelder 2005). The sustainability of the services wetlands provide are reliant on the integration of wetlands within a broader landscape (i.e., the configuration and composition of the landscape and connection to other water bodies) (Zelder, 2000; CEMA, 2013). However, this link between landscape and wetland-rich landscape reclamation is not regularly operationalized and standard height-to-length and length-to-width ratios could lead to the homogenization of wetlands throughout a region. Among different types of wetlands, permanent open-water wetlands will play an increasingly important role, particularly in Canada, due their increased creation and link to climate. To meet policy and regulatory objectives and reclaim these wetland landscapes to be integrated with the adjacent landscape and sustainable in the future, we must understand the landscape patterns surrounding permanent open-water wetlands and across the natural regions.

Our study analyzed 13, 676 spatially independent landscapes across three natural regions (Grassland, Parkland, and Boreal). A set of landscape metrics representing the spatial configuration of wetland landscapes was applied to a final subset of 1,684 permanent open-water landscapes and a set of non-parametric tests were used to determine if the landscape pattern of our sample landscapes differed across levels of disturbance and between natural regions. We found that the amount of permanent open-water in our landscapes was predominately low with ~85% of our samples having less than or equal to 8% (0.08 km^2) of the area classified as permanent open-water wetland.

Results identified four distinct landscape patterns, which varied across analysis at the inventory and natural region levels and identified limited landscape patterns in the low (0-20%) disturbance landscapes, with the exception being the Grassland region. These results indicate a reference condition approach may not be ideal for reclaiming permanent open-water wetland landscapes or that landscape patterns may have been affected by limited statistical power in some comparisons. Instead we recommend that climate envelopes containing similar proportions of permanent open-water wetlands be used as reference. Diversity of the land-cover was the exception, whereby it was statistically significantly different across all disturbance intervals and natural regions.

We had hoped to identify landscape patterns that would help identify the necessary landscape patterns under various amounts of disturbance to create sustainable permanent open-water wetlands, but the landscape patterns identified with this exploratory method supports the idea that permanent open-water wetlands are not as susceptible to disturbance as semi-permanent and temporary wetlands (Serran & Creed, 2014). Furthermore, the results presented in this exploratory research should not be interpreted as providing carte blanche to industry to develop with up to 80% disturbance without repercussion. These preliminary results indicate that permanent open-water wetlands alone can not be used to evaluate the effects of disturbance in wetland-rich landscapes, which function as spatially distributed complexes made up of several different types of wetlands (Zelder, 2000). There is a significant danger in focusing on reclaiming entire wetland-rich landscapes with a single type of wetland (i.e., permanent open-water wetlands), particularly in the Boreal region and given predicted climate changes (Schneider et al. 2015). To maintain diversity and sustainability of reclaimed wetland-rich landscape, regulators should enforce the creation of multiple wetland types (i.e., semi-permanent, temporary, seasonal) to maintain, diversity, functions and services.

The presented results were limited by the available data and would be advanced by a single, high-resolution dataset for the entire province that includes both wetland and land-cover data. While this would be a labour intensive and costly endeavour, it would provide a base to further investigate the effects of specific anthropogenic disturbances (i.e., roads, developed areas, types of agriculture) on permanent open-water wetland landscapes and other wetland types. A new dataset would have three basic requirements: 1) that the spatial extent be complete and represent a relatively homogeneous, climate region (i.e., natural region), 2) that a consistent temporal, thematic and spatial resolution be used in combination with a single methodology for both wetland classification, land-use and land-cover to ensure consistently among metrics and 3) that any new dataset strive for the highest level of accuracy possible for wetland delineation to provide the most accurate representation of landscape condition (Evans et al. 2017). To these suggestions, we add that any new classification systems used in the future attempt to identify and include man-made wetlands to ensure the integrity of the data for future wetland researchers.

Chapter 3 – Context and Future Directions

3.1. Context of Wetland Reclamation Within Landscape Ecology

Landscape ecology is focused on the interplay between spatial pattern and ecological processes and is widely recognized as an interdisciplinary science. The main objectives of landscape ecology are 1) to understand the relationships between ecological processes and spatial pattern, and 2) to use this understanding as a basis for landscape planning (Ahern, 1999; Wu & Hobbs, 2002). While we have made great strides in our understanding of landscape patterns and site-level ecological processes, integrating the concepts of ecological knowledge to landscape planning remains a challenge (Wu & Hobbs, 2002). Overcoming this challenge is imperative for successful landscape reclamation, particularly for large-scale mega-projects, as the integration of knowledge about ecological processes and spatial patterns into landscape planning will result in our ability to create sustainable landscapes that can maintain their landscape structure and ecosystem services over time (Opdam et al. 2006) and within a changing climate (Erwin,2009). To develop sustainable landscapes, we must aim for a landscape condition of stability in physical, ecological and social systems, while seeking to accommodate the needs of the present without compromising the ability of future generations ability to meet their needs (World Commission on Environment and Development, 1987; Ahern. 2002). In a world increasingly affected by anthropogenic disturbance, reclaiming landscapes to be sustainable, while maintaining their structure and ecosystems services will limit the burden on future generations.

Wetland-rich landscapes are comparable to coral reefs and rainforests in terms of their complexity and ecological importance (Costanza et al. 2014) and share a common threat from anthropogenic disturbance (Hansen et al. 2013; Moreno-Mateos et al. 2017). Due to the increasing awareness of their importance, wetlands have been the focus of many restoration (Zelder & Callaway, 1999; Zelder, 2000; Jessop et al. 2015; Baldwin et al. 2018) and reclamation efforts (Schipper & Reddy, 1994; Rooney et al. 2012; Roy et al. 2016) worldwide, however, our ability to successfully recreate these complex ecosystems has been limited (Mitsch & Wilson, 1996), both in terms of compliance (e.g., contract or permit) and functional (i.e., restored ecological function) success (Kentula, 2000).

The exploratory research presented in Chapter 2 furthers our understanding of permanent open-water wetland-rich landscapes and aligns itself with the first objective of landscape ecology, whereby landscape metrics were used to quantify the landscape pattern of permanent open-water wetland-rich landscapes across disturbance levels and spatially across three natural regions (Boreal, Parkland, and Grassland). To the best of our knowledge, the pattern of permanent open-water wetland-rich landscapes has not been quantified before and the results offer a benchmark for comparison to other similar regions and against other types of landscapes. The distribution of metric values can be used to inform landscape-level reclamation activities and is the first step toward establishing a link between pattern and function (i.e., landscape ecology objective 1).

While literature exists that establishes a relationship between landscape structure and anthropogenic disturbance (e.g., Zurlini et al. 2006; Luck & Wu, 2002; DiBari, 2007; Arroyo-Rodriquez et al. 2016; Matos et al. 2017), relatively little research has focused on this relationship in wetland-rich landscapes. A notable and recent exception is Evans et al. (2017), where metric values were analyzed across varying amounts of anthropogenic disturbance to show that low and highly disturbed wetland landscapes were significantly different. Although the presented methodology (Chapter 2) is similar, our methodology differs in that it focused on a specific subset of wetlands (permanent open-water), used an increased sample size of landscapes (3343 to 13, 676), accounted for variance, removed landscapes comprising 100% disturbance, and ensured samples were spatially independent by applying a 1000-meter minimum distance between landscapes (Kraft, 2016).

Despite the similarities between the presented research and that conducted by Evans et al. (2017) (i.e., study area, data, landscape metrics applied), results also differed. Wetland configuration and composition significantly differed between low and high disturbed landscapes in the study by Evans et al. (2017), while our focus on a specific subset (i.e., permanent open-water) showed that the configuration of permanent open-water wetland-rich landscapes were generally not as statistically significantly different when disturbance values were between zero and 80% , failing to identify a discernable landscape pattern in the low (0-20%) disturbance level (i.e., reference condition) except in the Grassland region. However, the exception across both studies, inventories, and natural regions was the composition or diversity of land-cover (i.e., SIDI_land, Chapter 2), which demonstrated significant metric values in every comparison of metric values across disturbance levels.

To guide landscape-level reclamation planning and closure, we interrogated permanent open-water wetland-rich landscapes along disturbance levels and across natural regions, our results identified four distinct landscape-patterns: 1) Boreal 0-80% disturbed, 2) Grassland 0-20% disturbed, 3) Grassland 0-80% disturbed, and 4) Boreal 80-99.9% disturbed. The absence of identifiable landscape patterns in the low (0-20%) disturbed landscapes in the Boreal and Parkland was interesting given that 30% of samples in the Central inventory and 18.9% of samples in the Southern inventory were in this lower disturbance interval. Essentially indicating that in place of a reference condition approach based on low disturbed permanent open-water landscape conditions, that disturbed landscapes containing similar amounts of permanent open-water, within their respective climate envelopes, can provide a preliminary and comparable pattern for future landscape-level reclamation efforts.

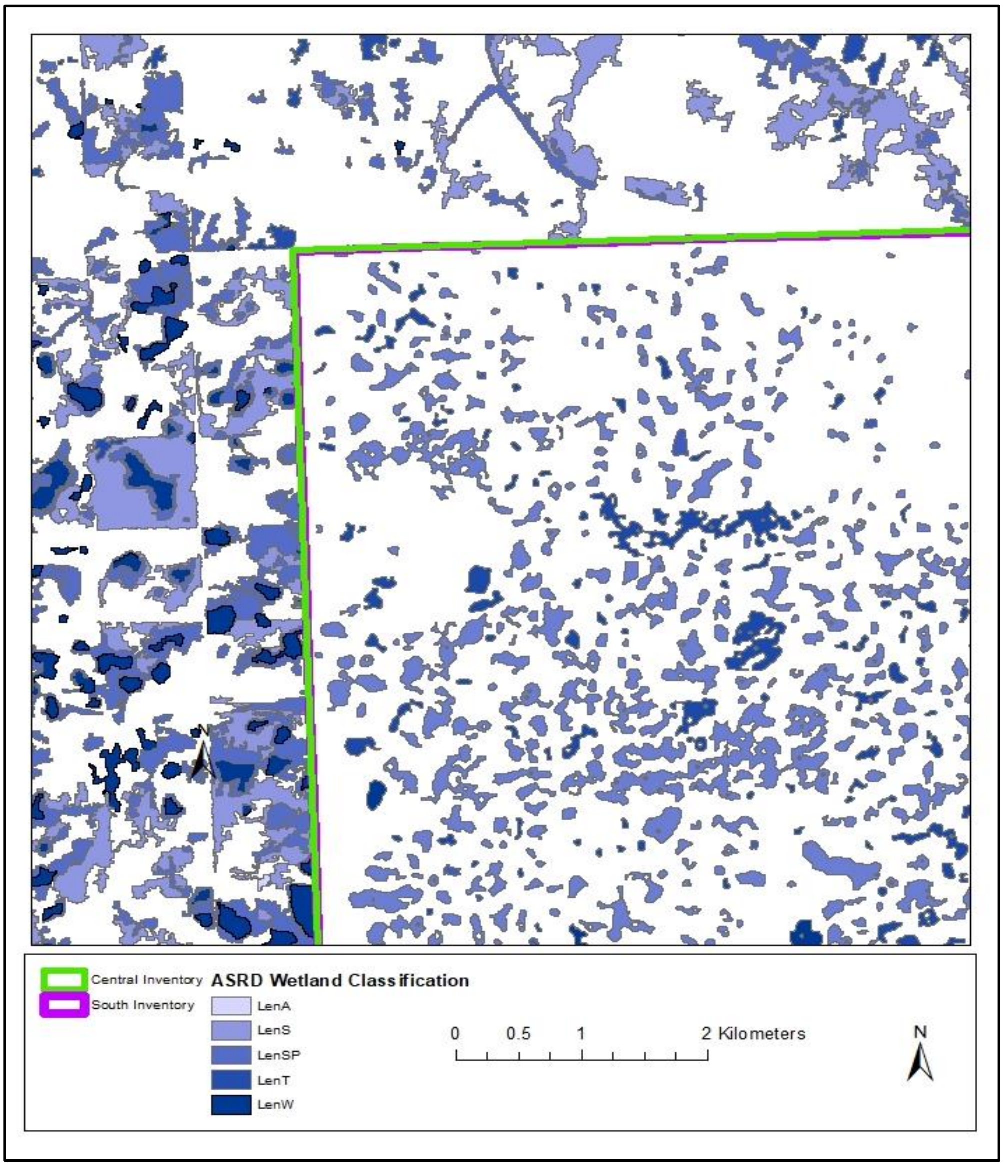
Aligning these results with the second objective of landscape ecology (i.e., using information obtained from the study of landscape pattern as a basis to inform landscape management), the four distinct landscape patterns identified (Chapter 2) may provide industry and regulators seeking to include permanent open-water wetlands in their closure plans with some general types of landscape structure to consider. However, based on the literature (e.g., Serran & Creed, 2014) and the limited landscape patterns identified in this exploratory research, we can conclude that permanent open-water wetlands alone are insufficient for capturing the landscape structure to maintain and create wetlands with varying amounts of disturbance.

Due to the high costs associated with integrating post-disturbance landscapes into the surrounding topography (Lemphers et al. 2010), landscape-level reclamation efforts concerning permanent open-water wetlands should seek to create an equivalent amount of permanent open-water found in natural landscapes. While landscapes in Alberta often contain permanent open water wetlands, we found that of the 1,684 open-water sample landscapes, the majority of these wetlands in both the Central and Southern inventories (86.1 and 85.3%) had less that or equal to 8% (0.08 km^2) of their total area classified as permanent open-water. Aligning this knowledge with Objective 2 (i.e., landscape ecology), this provides industry and regulators tasked with designing and reclaiming permanent open-water wetland landscapes with a preliminary guideline for the amount of permanent open-water needed per km^2 of reclaimed wetland-rich landscape, which can be used to gauge post-reclamation efforts. For example, if closure plans include permanent open-water wetlands above this threshold then novel ecosystems are being created that do not have natural equivalents in the any of the natural regions investigated in this research. If this is the case, future closure plans should be modified based on the range of metric values presented as part of this thesis and account for future climate to determine if these novel open-water wetlands can be sustained in the Boreal region.

Lastly, a brief analysis of the mean values of metrics associated with the composition and configuration of permanent open-water wetland-rich landscapes demonstrated that permanent open-water wetlands become less complex and elongated, and more compact and round when comparing permanent open-water wetland-rich landscapes from the Central to the Southern inventory, and from the Boreal to the Grassland regions.

This compaction and decreased shape complexity in permanent open-water wetlands towards the south, is likely a result of wetland drainage and consolidation, a common practice in the agricultural landscapes (McCauley et al. 2015), increased urbanization in Southern Alberta (Thom et al. 2001; Statistics Canada, 2016) and potentially by wetlands less than 0.2 ha being excluded by the minimum mapping unit in the Southern inventory. While these permanent open-water wetlands exist with high levels of urbanization and agriculture in their surrounding land-cover in both inventories, consolidation usually involves temporary and semi-permanent wetlands being combined into larger open-wetland systems. This leads to higher water levels that alter the productivity and function of these ecosystems (McCauley et al. 2015). For example, wetlands that exist in areas that have been extensively drained and consolidated have larger surface areas, dry out less frequently, and have increased surface connectivity to surrounding wetlands through ditches. These conditions are more favourable for fish, which decreases the presence of invertebrates and ultimately reduce the overall productivity and quality of these wetlands for many species, including breeding waterfowl that rely heavily on impermanent wetlands in Alberta (McCauley, et al. 2015). Therefore, we should consider building complexes of open-water wetlands and included impermanent wetlands types to mimic the natural heterogeneity of wetland-rich landscapes and can retain productivity and diversity in reclaimed landscapes.

Contrasting results for the Parkland region (i.e., Central Parkland and Southern Parkland) demonstrate sensitivity of landscape metrics to data quality and possibly sample size. While the Central inventory contains larger polygons and a greater diversity of wetland classes, some differences are likely due to the differing spatial extents (i.e., Boreal and Parkland covered by Central; Parkland and Grassland covered by Southern). However, when looking at the differences in neighbouring boundaries between the inventories, methodological differences in their creation also play a role (Figure 3.1). As a result, sample landscapes within different inventories were analyzed separately and contrasting results were seen in the Parkland region. This suggests further analysis is needed regarding the effects of resolution, minimum mapping units, wetland delineation and classification techniques, and the spatial extent of sample landscapes on our ability to distinguish statistically significant differences using separate inventories.



3. 1 Example of the effects of different methodologies used in creation of the wetland inventories. Central inventory is top left, Southern inventory is the bottom right

The increased accuracy, statistical power and inclusion of wetlands smaller than 0.2 ha associated with the Central inventory produced significant metric values across disturbance intervals that were not seen in the Southern Parkland subset. While much of this can be attributed to data quality, the lack of data for the central region of the Parkland in either inventory was also a factor (Figure 2.1, Chapter 2). Further, while the Parkland region is an ecotone between the Boreal and Grassland, we can infer that the characteristics of the northern part of the Parkland region are similar to those of the southern Boreal region, as is the southern Parkland similar to the Grassland region. These similarities and data issues are likely a contributing factor to the lack of significant results for the Parkland region in both inventories.

2. Future directions

Chapter 2 made new contributions to landscape ecology and our understanding of permanent open-water wetland-rich landscapes by exploring landscape patterns across three natural regions (Boreal, Parkland, and Grassland). This exploratory research increased our understanding of the potential ability to identify the landscape patterns associated with permanent open-water wetlands using landscape metrics and disturbance levels. However, some questions remain from Chapter 2 whose answers would better inform suggestions for future directions in landscape ecology and permanent open-water wetland landscape reclamation.

2.1 Comparing LenW to LenSP and LenT

To understand the lack of identifiable landscape pattern in low (0-20%) disturbance permanent open-water landscapes in the Boreal and Parkland regions and the contrasting results between inventories for the Parkland region, we applied the same methodology from Chapter 2, to two other wetland subsets in the original dataset: semi-permanent and temporary lentic wetlands in both inventories (Central and Southern). Results indicate that these more impermanent wetlands types had statistically different landscape patterns between the low (0-20%) and high (80-99%) disturbance landscapes, as well as some distinction among the moderately disturbed landscapes (20-60%).

By applying the same methodology from Chapter 2 to lentic wetland subsets for semi-permanent (LenSP) and temporary (LenT), the results of the pairwise comparisons demonstrated that metric values in the low (0-20%) disturbance and high (80-99.9%) disturbance landscapes were statistically significant from the rest of the disturbance intervals in 9 of the 12 subsets, with the exceptions being LenSP Southern All (Appendix 22), Southern LenSP Parkland (Appendix 24), and LenT Grassland (Appendix 30). Limited sample size and statistical power may have played a role in the LenSP Southern All and LenSP Parkland subsets (Table 3.1); however, the large sample size in the LenT Grassland identified a less systematic landscape pattern given 14 of 36 metrics were identified as statistically significant in the low disturbance landscapes and 16 of 36 in the high disturbance landscapes (Table 3.1; Appendix 30).

Sample sizes in the Southern Parkland for LenSP and LenT were less than those in Central Parkland (Table 3.1), as was the case for LenW (Table 2; Chapter 2), and where sample sizes were higher (i.e., LenT Southern Parkland; Table 1) low (0-20%) disturbed landscapes were significant different from the other disturbance intervals, but no systemic landscape structure was seen in the high (80-99.9%) disturbed landscapes (Appendix 29). Large samples are often preferred in any analysis, because when all other things are equal, larger sample sizes tend to maximize the accuracy of population estimates, increase the generalizability of results, and decrease the probability of errors (Osborne & Costello, 2004). While it is likely that increased sample size in the Southern inventory would improve our ability to identify distinct landscape structures in the LenSP subsets, contrasting results were observed in the Parkland region for both LenSP (Appendix 24) and LenT (Appendix 29), the latter of which had more than double the amount of landscape samples, highlight that fact that data quality is affecting our ability to identify consistent landscape structure in the Parkland region between the Central and Southern inventories.

3. 2: Landscape samples containing LenSP and LenT wetlands in this research.

Inventory	Total	Total LenSP	Total LenT	Natural Region	LenSP	LenT
Central	4597	2996	3015	*Boreal*	1916	1944
				Parkland	1080	1017
Southern	9079	686	3618	*Parkland*	104	588
				Grassland	582	3030

From this additional analysis, we confirmed that permanent open-water wetlands are less susceptible to disturbance and that less permanent wetlands may offer a better representation of the effects disturbance on the structure of wetland-rich landscapes. By investigating less permanent wetland types (LenSP and LenT), we found them to generally have statistically significant landscape structures between low (0-20%) and highly (80-99.9%) disturbed landscapes, and in some cases (e.g., Central Boreal) moderately disturbed (60-80%) landscapes (Appendix 14).

2.2 The relationship between open-water wetlands and topography

The majority of open-water wetlands in our study area are considered geographically isolated wetlands (GIW) and are found within a terrain that was carved out by an extensive history of glaciation (Johnson et al. 2005). Terrain analysis has been useful in the quantification of landscape characteristics and establishing a relationship to site-properties. For example, topography metrics have been used to predict wetland location and frequency of inundation (Lang et al. 2013), which highlights that topography plays an important role in the spatial arrangement of wetlands. Considering the history of glaciation in the region and the role topography plays in the location and water balance of wetlands, the relationship between permanent open-water and topography may be stronger than the effects of disturbance in samples containing less than 80% disturbance. However, it has been shown that in post-disturbance mining landscapes, wetlands are less connected hydrologically and more susceptible to disturbance due to the thinner surficial geographic layers in these landscapes (Kompanizare, et al 2018), which suggest the need to compare permanent open-water wetlands in non-mining and mining landscapes to evaluate this hypothesis. To accomplish this a more consistent and complete dataset would be required.

2.4 The need for better data and the inclusion of man-made wetlands

Having a more consistent and complete dataset would not only increase the integrity of our results, it would also provide the opportunity to explore and compare the effects of specific disturbance types (i.e., agriculture, mining, development, roads) on the landscape patterns of open-water wetlands, and any other subset of lentic wetlands. For example, disturbances such as roads and bridges can directly change the slope and hydrological connectivity of surface-water habitats and unpaved roads can increase sediments into wetlands, increasing turbidity and reducing productivity (Trombulak & Frissell, 2000). Additionally, research has found that the diversity of herptile species in wetlands declines based on the density of roads within a 2 km perimeter (Findlay & Houlahan, 1997). The ability to analyze specific disturbances on the landscape structure of wetlands may be useful for pre-approval planning and help industry reduce their disturbance footprint before they even break ground. Lastly, reclaiming wetlands will take time, therefore having a quality dataset that classifies man-made wetlands will increase the ability of landscape managers to monitor and evaluate success of landscape-level wetland reclamation efforts under a changing climate.

While data quality can always be improved, this research demonstrates our ability to analyze separate datasets and identify landscape patterns across three natural regions, which can be used as preliminary step towards the creation of criteria and guidelines for the certification of landscape-level wetland reclamation that accounts for the cumulative effects within the landscape. As data quality improves, the methodology used in this thesis will still be applicable.

7. Chapter 1 References

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Appendix 1: Description of Natural Regions

Natural Regions are the largest mapped ecological units in Alberta's classification system. These units are defined based on landscape patterns such as soils, vegetation, and physiographic features. Their distribution is the result of influences from topography, climate and geology (Downing & Pettapiece, 2006)

Natural Region	% of Province	Natural Subregions	Wetlands & Water	Land-use
Rocky Mountain	6	Alpine	Wetlands uncommon, 4% glaciers	Conservation and recreation
		Subalpine	2% wetlands, 1% lakes & streams	Recreation, forestry, oil & gas, coal mining, minimal grazing
		Montane	2% wetlands, 1% lakes & streams	Recreation, forestry, grazing, major transportation corridors
Foothills	10	Upper Foothills	10% wetlands (valleys), <1% lakes & streams	Recreation, oil & gas, coal mining, forestry, minimal grazing
		Lower Foothills	20% wetlands (valleys), <1% lakes & streams	Recreation, oil & gas, coal mining, forestry, grazing, till cropping
Grassland	14	Dry Mixedgrass	3% wetlands /marshes or temporary; 2% lakes & streams	Oil & gas, grazing, irrigation farming
		Mixedgrass	5% wetlands, mainly marshes);1% lakes & streams	Oil & gas, grazing, irrigation farming
		Northern Fescue	7% wetlands, mainly marshes; 3% (lakes & streams)	Oil & gas, grazing, irrigation farming
		Foothills Fescue	3% wetlands, 1% lakes & streams	Recreation, oil & gas, grazing, till cropping
Parkland	9	Foothills Parkland	4% wetlands, <1% lakes & streams	Oil & gas, grazing, till cropping
		Central Parkland	10% wetlands, mainly marshes, 2% lakes & streams	Oil & gas, grazing, till cropping
		Peace River Parkland	6% wetlands, 2% lakes & streams	Oil & gas, grazing, till cropping
Boreal Forest	58	Dry Mixedwood	15% wetlands, 3% lakes & streams (not including Slave lake)	Forestry, oil & gas, coal mining, recreation, grazing and till cropping in south
		Central Mixedwood	40% wetlands (mainly peat); 3% lakes & streams	Forestry, oil & gas, coal mining, recreation, grazing, till cropping in south, subsistence
		Lower Boreal Highlands	30% wetlands (Chinchaga area); 1% lakes & streams	Forestry, oil & gas, recreation, subsistence
		Upper Boreal Highlands	35% wetlands, 1-2% lakes & streams	Forestry, oil & gas, recreation, subsistence
		Athabasca Plain	20% wetlands, 3% lakes & streams	Recreation, subsistence
		Peace-Athabasca Delta	20% wetlands, 40% shallow lakes and channels	Recreation, subsistence
		Northern Mixedwood	70% wetlands, 3% lakes & streams	Forestry, oil & gas, recreation, subsistence
		Boreal Subarctic	60% wetlands, 2% lakes	Oil & gas, fishing
Canadian Shield	1	Kazan Upland	20% wetlands, 10% lakes	Mineral extraction, recreation, subsistence

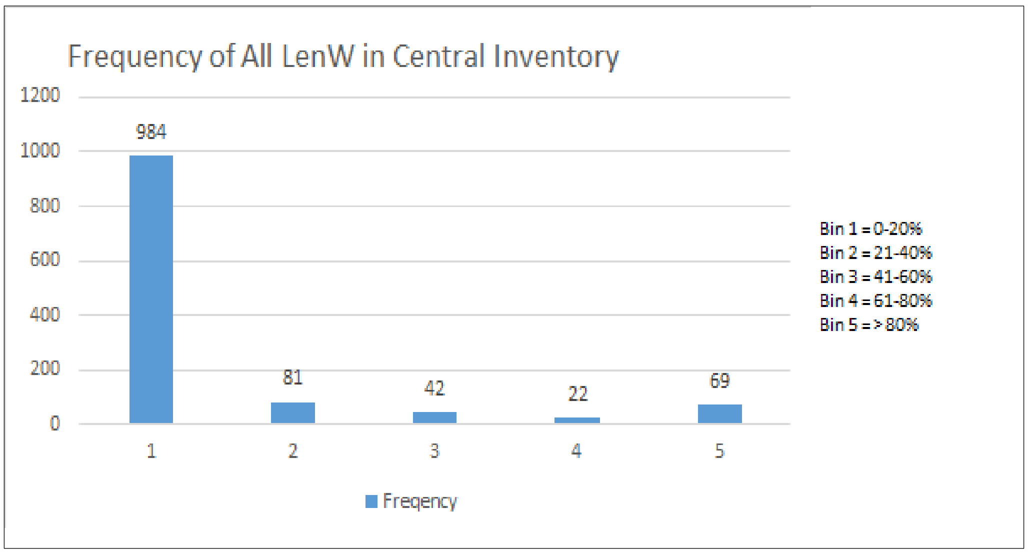
Appendix 2: AAFC land-cover classification

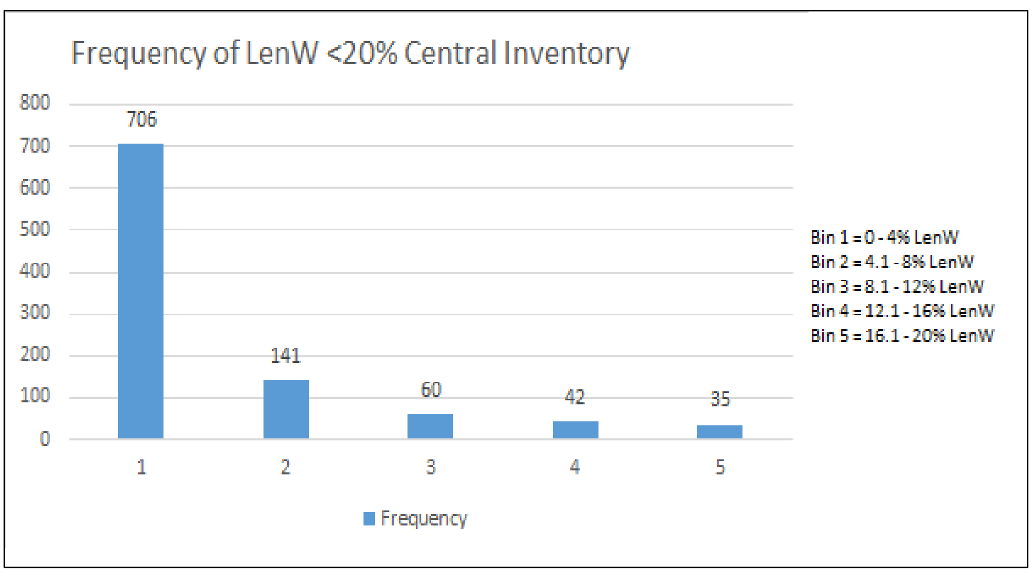
Landcover	Description
Water	Waterbodies (lakes, rivers, streams, reservoirs, salt water, etc.).
Exposed	Naturally occurring but predominately non-vegetated and undeveloped (glacier, rock, sediments, burned areas etc.)
Developed	Built-up land that includes roads, buildings, urban, paved and industrial sites etc.
Shrub	Mostly woody vegetation <2 m in height. Includes grass or wetlands with woody vegetation, new forests etc.
Wetland	Land with a high-water table that is near or above the soil surface for long enough to promote wetland processes.
Grassland	Predominately native grasses and other herbaceous vegetation.
Agriculture	Annual crops, perennial crops, pasture, excludes native grasses
Forest	Forested or treed areas > 2m. Incudes deciduous and coniferous.

*Agriculture and developed land-cover classes were used to calculate the amount of disturbance in the sample landscapes

Appendix 3: Preliminary data investigation of LenW in the inventories

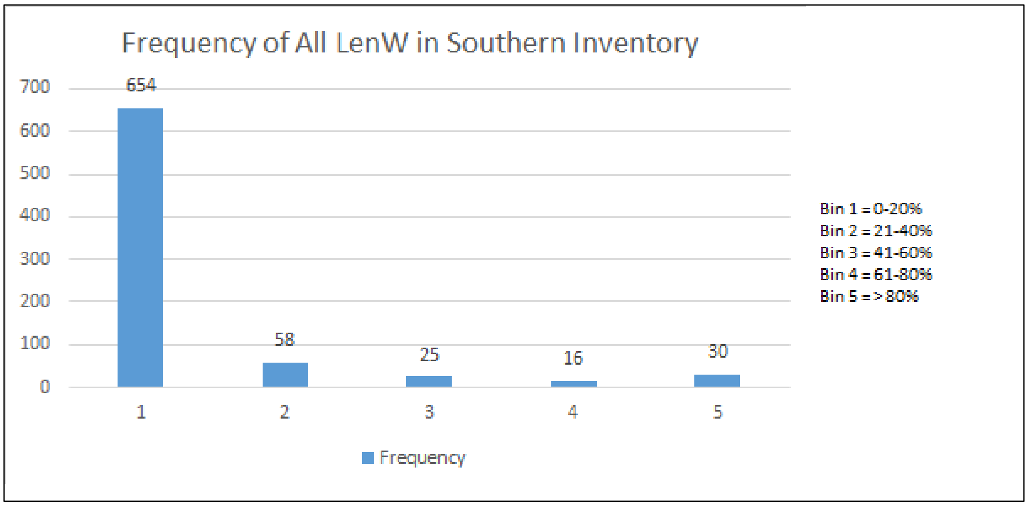
From the original 4597 random landscapes in the Central inventory, 1198 had LenW ASRD wetlands in the samples. Frequency plots highlight that that 81.82% of LenW landscapes were less than or equal to 20% of the sample landscapes for both inventories. Analysis of landscapes containing LenW >20% showed a high mis-classification rate (93.33%) when a random sample (n=30) was inspected visually. Therefore, we decided to focus on the 984 LenW samples less than or equal to 20% of the landscape. In this new subset (LenW20), 71.75% of LenW wetlands were less than 4% of the samples in all the samples and 86% were less than 8% of all the samples. This trend was also found during the analysis of the Southern Inventory.

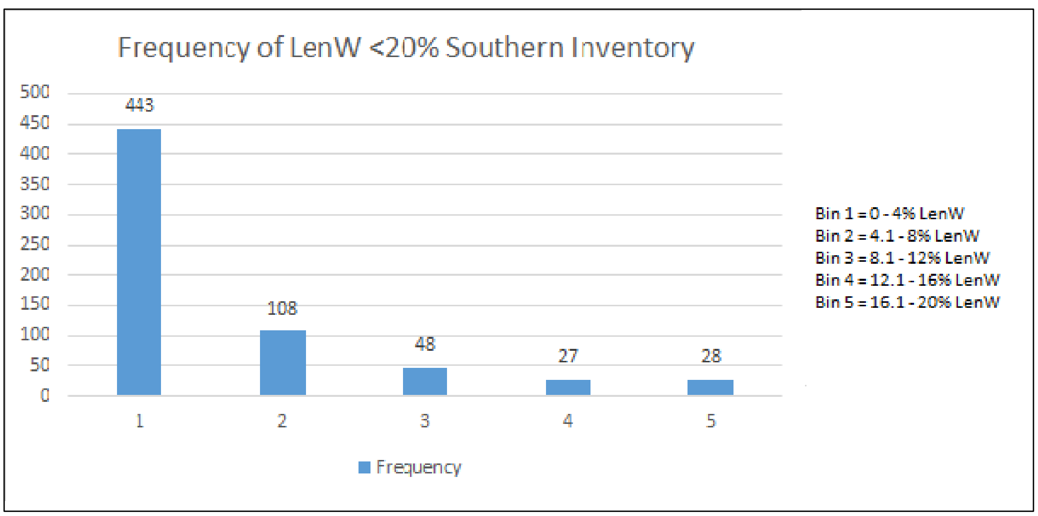




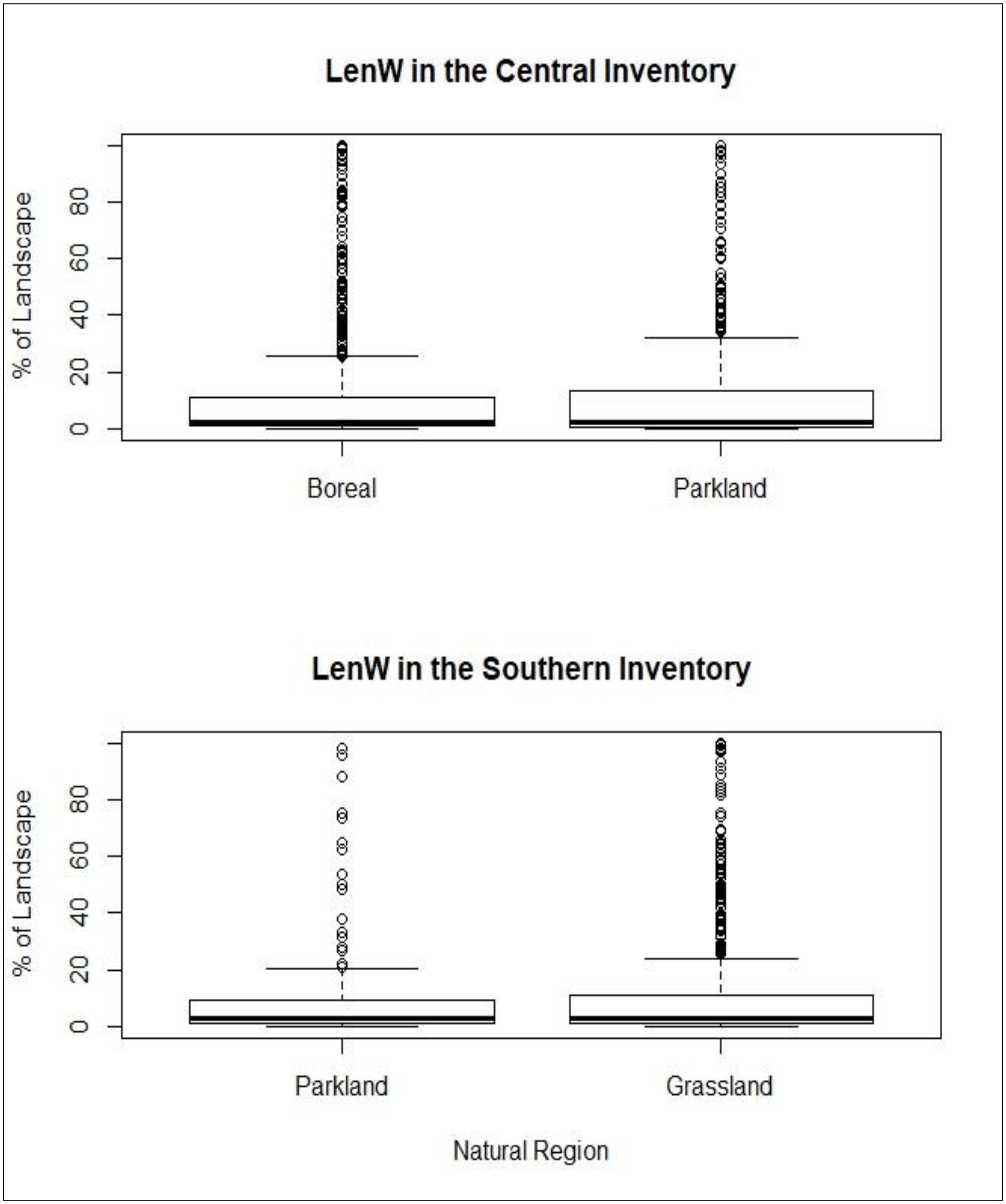
Appendix 3 continued

From the 9079 random sampled landscapes in the Southern Invnetory, 783 had LenW in them. 83.5% of all the samples in the southern inventory have less than 20% permanent open-water in them. Of those 654 samples, 67.74% were less than 4% of the landscape and 84.2% of landscapes had less than 8% LenW.



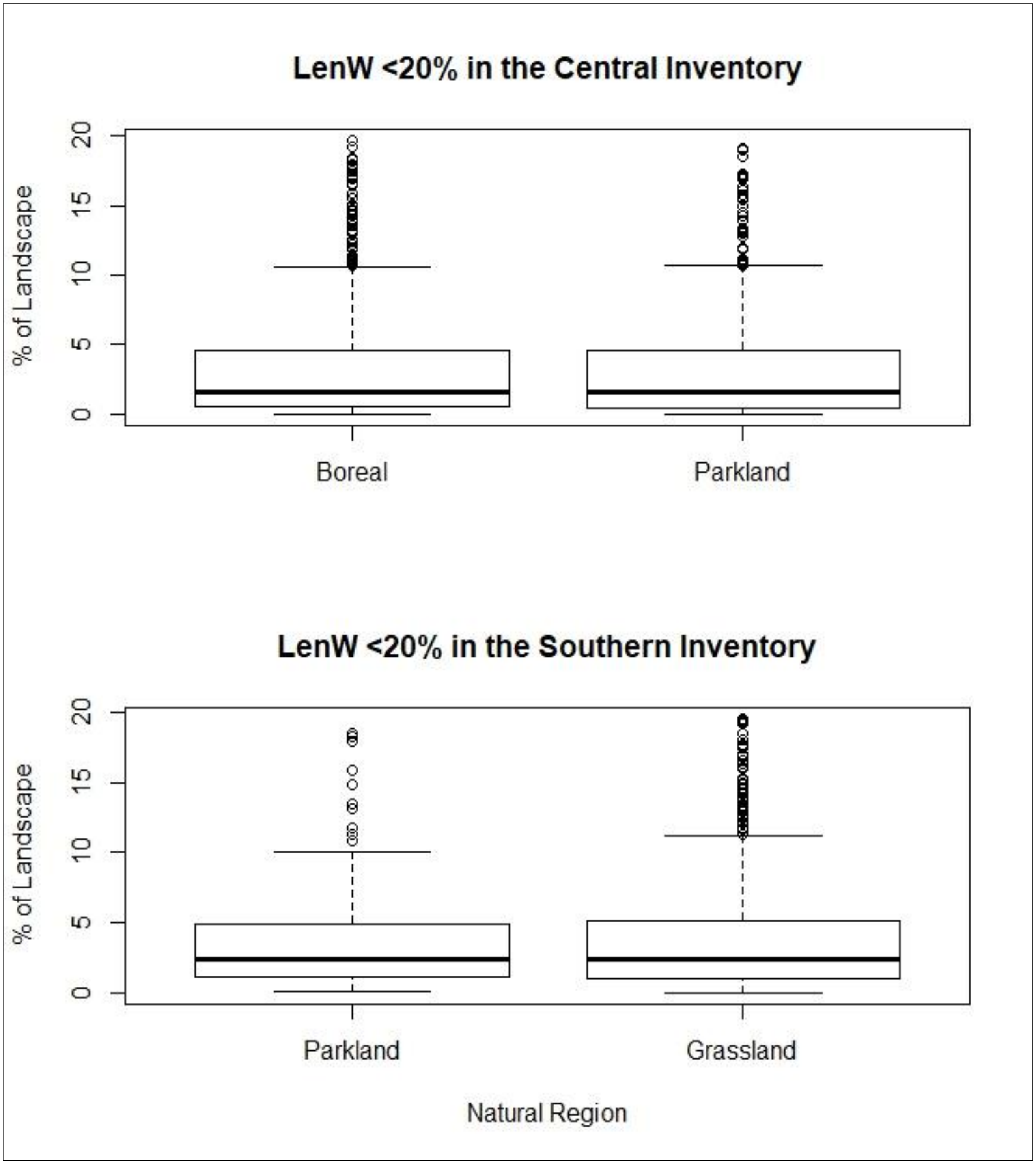


Appendix 4 Boxplots of the percent LenW in the samples



Appendix 4 continued

Boxplots of the percent LenW20 (less than or equal to 20%) of the total sample landscape.



Appendix 5: Description of wetland landscape metrics (Adapted from McGargial et al. 2012).

Type	Metric	Abbr.	Formula	Units	Range	Description
Shape	Area-weighted mean shape index	SHAPE_AM	$SHAPE = \frac{p_{ij}}{\min p_{ij}}$	NA	SHAPE ≥ 1	SHAPE= patch perimeter (given in number of cell surfaces) divided by the minimum perimeter (given in number of cell surfaces) possible for a maximally compact patch (in a square raster format) of the corresponding patch area. SHAPE = 1 when the patch is maximally compact (square or almost square) and increases w/o limit as shape becomes more irregular. Straightforward measure of overall shape complexity.
	Area-weighted mean related circumscribing circle	CIRCLE_AM	$CIRCLE = 1 - \left[\frac{a_{ij}}{a_{ij}^{s}}\right]$	NA	0 ≤ CIRCLE ≤ 1	CIRCLE= 1 minus patch area (m^2), divided by the area (m^2) of the smallest circumscribing circle. Uses smallest circumscribing circle instead of square despite the raster format. CIRCLE =0 for circular patches and 1 for elongated. Useful for distinguishing between patches that are both linear and elongated.
Aggregation	Aggregation index	AI	$AI = \left[\frac{g_{ii}}{\max \rightarrow g_{ii}}\right](100)$	%	0 ≤ AI ≤ 100	AI is calculated from an adjacency matrix and equals the number of like adjacencies involving the focal class, divided by the maximum possible number of like adjacencies involving the class. AI does not account for adjacencies involving other

						classes. It is tallied using a single count method where each cell side is only counted once. AI is range between 0 and 100% where 100 represents a patch that is maximally aggregated into a single compact patch
	Contagion index	CONTAG	$CONTAG = \left[1 + \frac{\sum_{i=1}^{m}\sum_{k=1}^{m}\left[(P_i)\left(\frac{g_{ik}}{\sum_{k=1}^{m} g_{ik}}\right)\right] \cdot \left[\ln(P_i)\left(\frac{g_{ik}}{\sum_{k=1}^{m} g_{ik}}\right)\right]}{2\ln(m)}\right](100)$	%	0 < CONTAG ≤ 100	Contagion can be described as the observed contagion over the maximum possible contagion for the given number of patch type (%). It is inversely related to edge density. When edge density is low, the class occupies a very large proportion and contagion is high, and vice versa. Contagion is affected by both the dispersion and interspersion of patch types. Low levels of patch dispersion (i.e., high proportion of like adjacencies) and low-levels of patch type interspersion. CONTAG =100 when a patch is maximally compact.
	Patch cohesion index	COHES	$COHESION = \left[1 - \frac{\sum_{i=1}^{m}\sum_{j=1}^{n} p_{ij}}{\sum_{i=1}^{m}\sum_{j=1}^{n} p_{ij}\sqrt{a_{ij}}}\right]\left[1 - \frac{1}{\sqrt{A}}\right]^{-1} \cdot (100)$	%	0 < COHES < 100	Patch cohesion index is a measure of the physical connectedness of the corresponding patch type. Patch cohesion increases as the patch type becomes more clumped or aggregated in its distribution; hence, more physically connected it is. * Behaviours of this metric at that landscape level has not been evaluated

	Area-weighted mean Euclidean nearest neighbour	ENN_AM	$$\mathrm{ENN} = h_{ij}$$	Meters	ENN > 0	ENN equals the distance (m) to the nearest neighbouring patch of the same type and is based on the shortest edge-to-edge distance. It a simple measure of patch context and is used extensively to quantify patch isolation. Uses nearest neighbour distance as defined by Euclidean geometry to be the shortest straight-line between focal patches.
	Splitting index	SPLIT	$$\mathrm{SPLIT} = \frac{A^2}{\sum_{i=1}^{m}\sum_{j=1}^{n} a_{ij}^2}$$	NA	1 ≤ SPLIT ≤ (Number of cells in landscape)	SPLIT is based on the cumulative patch area distribution and is interpreted as the effective mesh number or number of patches with a constant size when the landscape is subdivided into S patches, where S is the value of the splitting index. It is intended to be a measure of fragmentation in a landscape where SPLIT= 1 consists of a single patch.
Diversity	Simpson's diversity index	SIDI	$$\mathrm{SIDI} = 1 - \sum_{i=1}^{m} P_i^2$$	NA	0 ≤ SIDI < 1	SIDI is a popular measure of diversity borrowed from community ecology. Simpson's is less sensitive to the presence of rare types and it's a range from 0 to 1, where SIDI= 0 contains only one patch. As SIDI approaches 1, patch richness increases and the proportional distribution of area among patch types becomes more equitable.

$\boldsymbol{a_{ij}}$ = 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*

gii = number of like-adjacencies for pixels of class i

gij = number of adjacencies between pixels of classes i and j

$\boldsymbol{h_{ij}}$ = distance from patch *ij* to nearest patch of the same class (edge to edge distance)

m = number of classes

$\boldsymbol{n_i}$ = number of patches for class *i*

$\boldsymbol{p_{ij}}$ = perimeter of patch *ij*

Pi = proportion of landscape occupied by class i5

Z = number of cells in the landscape

For more information about landscape metrics calculated in Fragstats:

https://www.fs.fed.us/pnw/pubs/pnw_gtr351.pdf

https://www.umass.edu/landeco/research/fragstats/documents/fragstats.help.4.2.pdf

Appendix 6: Statistical Analysis

The Shapiro-Wilk's test is used to determine if a random sample comes from a normal distribution. The test gives you a *W* value, where small values allow you to reject the null hypothesis that the data is normally distributed based on a threshold (e.g., p-value <0.05) (Shapiro & Wilk, 1965). In our study, non-normality of the data was revealed in every subset.

$$W = \frac{\left(\sum_{i=1}^{n} a_i x_{(i)}\right)^2}{\sum_{i=1}^{n} (x_i - \bar{x})^2},$$

Q-Q plots or quantile-quantile plots are a graphical representation of distribution that is used to test for normality. Two sets of quantiles are plotted against each and if both sets of quantiles come from the same distribution, the graphically representation will be nearly straight (Wilk & Gnanadesikan, 1986). This graphically analysis of our data further confirmed non-normality.

The Brown-Forsythe test is a modification of the Levene's test that uses medians instead of means to test for the assumption of equal variance (Brown & Forsythe, 1974).

$$F = \frac{(N-p)}{(p-1)} \frac{\sum_{j=1}^{p} n_j (\tilde{z}_{\cdot j} - \tilde{z}_{\cdot\cdot})^2}{\sum_{j=1}^{p} \sum_{i=1}^{n_j} (z_{ij} - \tilde{z}_{\cdot j})^2}$$

The Kolmogorov-Smirnov is a non-parametric test that was used on metrics that had unequal variances. This was done both within (one-sample) and between (two-sample) groups (Darling, 1957) and manually scripted into a pairwise comparison.

$$F_n(x) = \frac{1}{n} \sum_{i=1}^{n} I_{[-\infty, x]}(X_i)$$

The Kruskal-Wallis (K-W) test is a non-parametric test that quantitatively compares the metric distribution among the disturbance intervals and provides a measure of stochastic dominance among groups (Kindscher, Fraser, Jakubauskas, & Debinski, 1998; Kruskal & Wallis, 1952). The K-W test is similar to an analysis of variance (ANOVA) except it does not assume normal distribution, however, it does require that the variance be similar among the groups (Elliott & Hynan, 2011) and so it was used on metrics with equal variance. H is calculated as

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

where N is the number of observations in all samples combined, Ri is the sum of the ranks in the sample *i, ni* is the number of observations in sample *i*, and *C* is the number of samples. In cases where results are equal or tied, each observation is given the mean rank of the tie and Equation (1) is modified to

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

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

$$T = t3 - t$$

and *t* is the number of tied observations in the group. H follows a chi-squared distribution where higher values express the difference between at least two of the groups assessed is statistically significant (Kruskal & Wallis, 1952).

The accompanying p-values along the chi-square distribution specify the probability that the median ranks of the groups are the same. The K-W test is a useful omnibus test, but it does not provide the specific group that is significantly different from each other group. As such, the Dunn's post-hoc was required where significant differences occurred (Dunn, 1964). The Dunn's test uses a z-score that is calculated between the mean ranks of the two groups being compared (Dinno, 2015). The z-score for comparing groups A and B is calculated as

$$zAB = yAB/\ \sigma AB$$

Where yAB is the difference in mean ranks for groups *A* and *B*, and σAB are the standard error of yAB that is calculated as

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

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.

In summary, metric distributions were compared using boxplots, normality was tested both quantitively with the Shapiro-Wilks test and graphically with Q-Q plots. Variance was tested with the Brown-Forsythe test and where metric values had unequal variance, the Kolmogorov-Smirnov test was conducted within and between groups and manually computed into a pairwise comparison. Where metric values had equal variance, the Kruskal-Wallis tests was used and followed with a Dunn's pairwise comparison. Results for all subsets were combined into a final pairwise table to identify landscape pattern.

Appendix 6 Continued

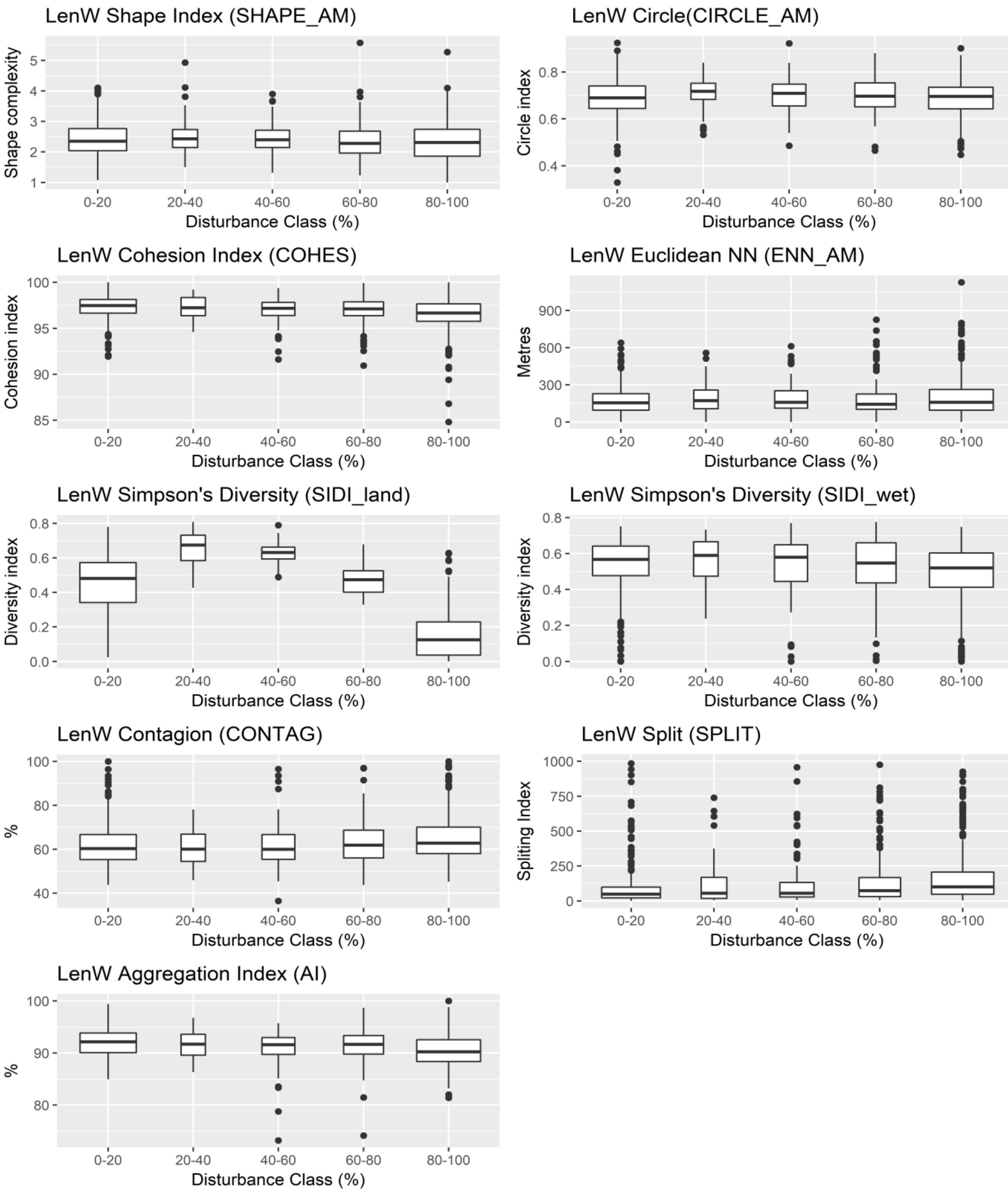
Statistical power in each subset and disturbance interval. All calculations were done with an alpha of 0.05 (confidence level of 95%.) Confidence intervals (CI) were also reported on the pairwise comparison tables included throughout this document.

Statistical Power						**Disturbance Interval CI**				
Inventory	**Sample Size (*n*)**	**Confidence Interval (CI)**	**Natural Region**	**Sample Size(n)**	**Confidence Interval (CI)**	**0-20**	**20-40**	**40-60**	**60-80**	**80-100**
Central	984	3.17				n = 295 CI = 5.71	n = 56 CI = 13.1	n = 102 CI = 9.7	n = 149 CI =8.03	n = 382 CI = 5.01
			Boreal	*632*	3.9	n = 188 CI = 7.15	n = 38 CI =15.9	n = 59 CI =12.76	n = 98 CI = 9.9	n = 249 CI =6.21
			Parkland	*352*	5.22	n = 107 CI = 9.47	n = 18 CI = 23.1	n = 43 CI = 14.49	n = 51 CI = 13.72	n = 133 CI = 8.5
Southern	700	5.22				n = 132 CI = 8.53	n = 64 SI = 12.25	n= 69 CL = 11.8	n = 117 CI = 9.06	n = 318 CI = 5.5
			Parkland	*141*	8.25	n = 22 CI = 20.89	n = 17 CI = 23. 77	n = 14 CI = 26.19	n = 16 CI = 24.5	n = 72 CI = 11.55
			Grassland	*559*	4.14	n = 107 CI = 9.47	n = 18 CI = 23.1	n = 43 CI = 14.94	n = 51 CI = 13.72	n = 133 CI = 8.5

All calculations done with an alpha of 0.05/ confidence level of 95%

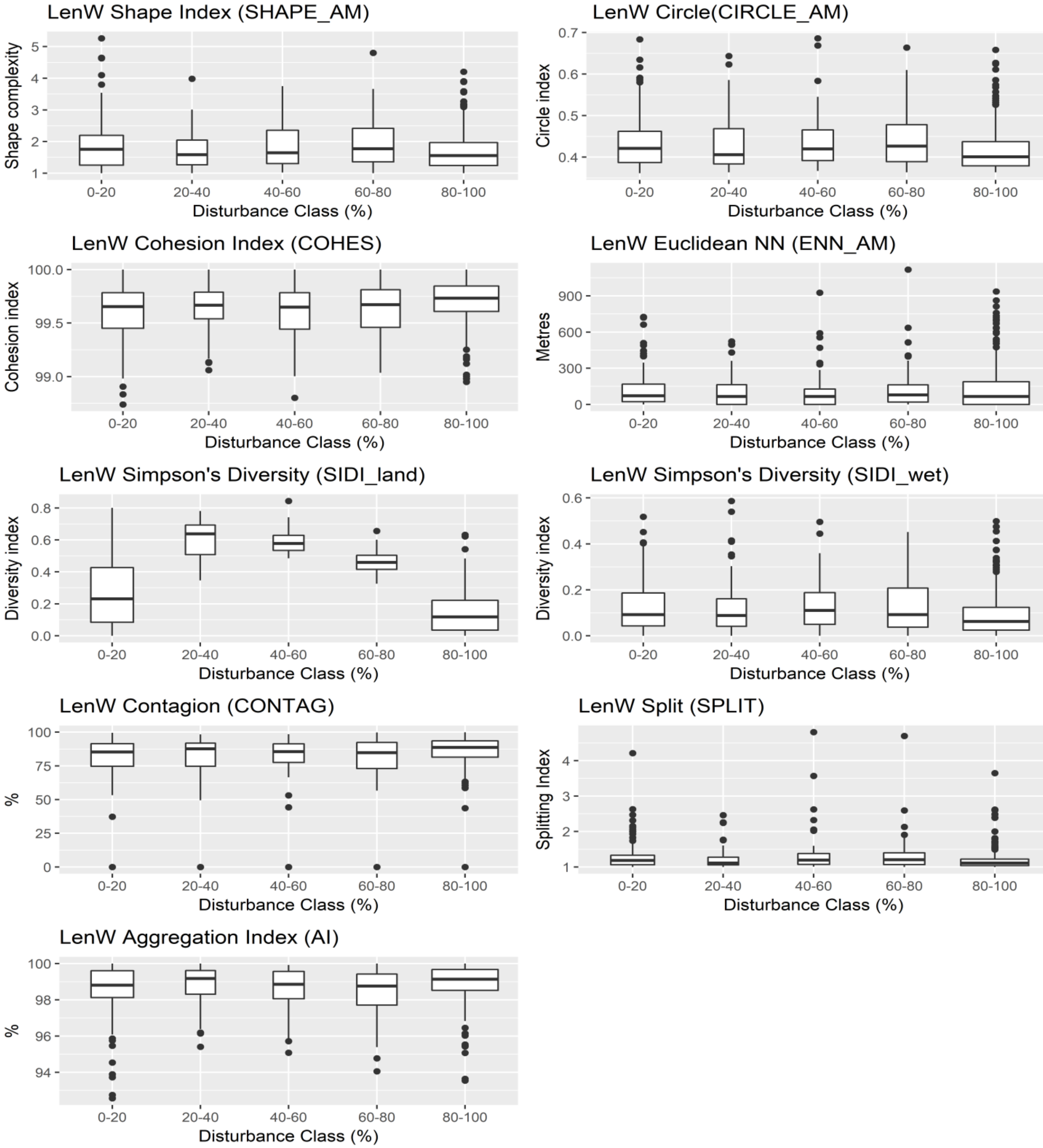
Appendix 7: Boxplots for the Central and Southern inventories

Distributions of the nine metrics for the Central inventory with natural regions combined.



Appendix 7 continued

Distributions of the nine metrics for the Southern inventory with natural regions combined.



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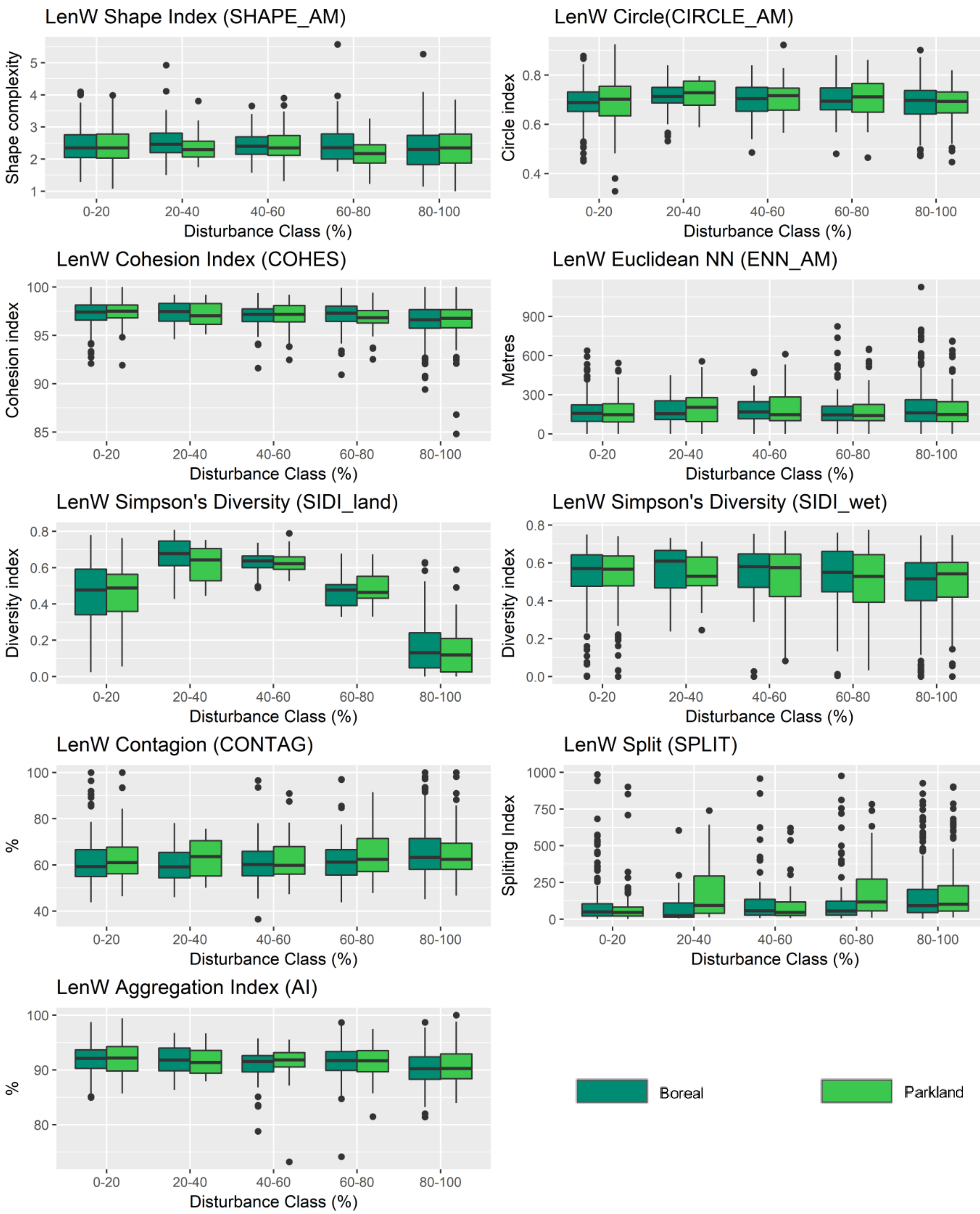
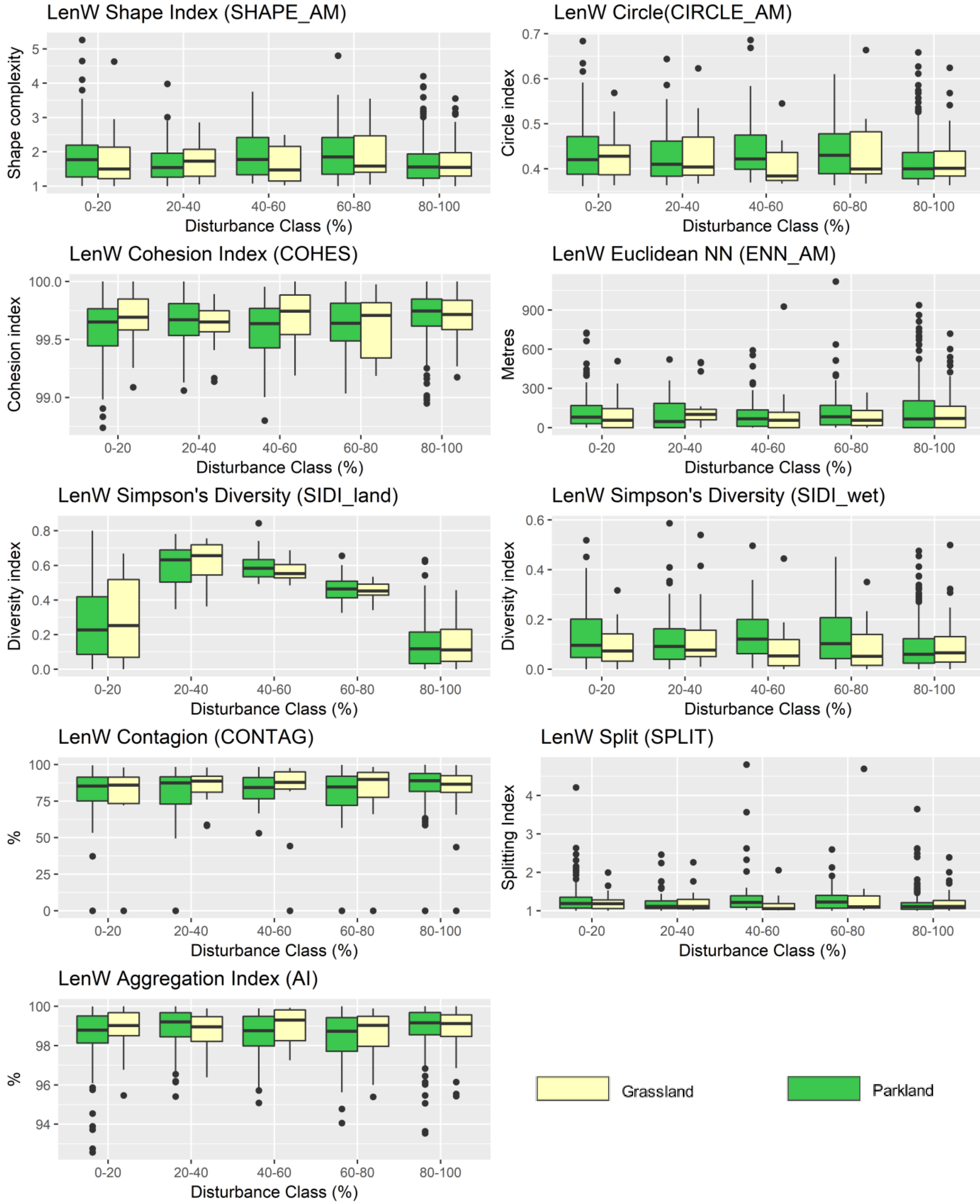
Appendix 8: Boxplots for the Central and Southern inventories split by natural regions

Distributions of the nine metrics for Central Inventory separated by natural region.



Appendix 8 continued

Distributions of the nine metrics for the Southern inventory separated by natural region.



Appendix 9: Kruskal-Wallis and Kolmogorov-Smirnov test results for equivalent disturbance levels in each inventory

Significance levels (p-values) of the Kruskal-Wallis and Kolmogorov-Smirnov comparisons of metric distributions between Natural Regions (Boreal and Parkland) in the Central inventory of equivalent disturbance levels.

		Disturbance (%)				
Type	**Metric**	**0-20**	**20-40**	**40-60**	**60-80**	**80-100**
Shape	SHAPE_AM	0.51	0.219	0.86	**0.016**	0.699
	CIRCLE_AM	0.536	0.605	0.694	0.452	0.652
Aggregation	AI	0.533	0.792	0.397	0.863	0.563
	COHESION	0.246	0.599	**<0.001**	0.097	0.743
	CONTAG	0.246	0.262	0.697	0.126	0.363
	ENN_MN	0.76	0.752	0.911	0.978	0.565
	SPLIT	0.595	0.136	0.638	**<0.001**	0.429
Diversity	SIDI_wet	0.401	0.243	0.609	0.152	0.311
	SIDI_land	0.967	0.051	0.5	0.253	0.124

Significance levels (p-values) of the Kruskal-Wallis and Kolmogorov-Smirnov comparisons of metric distributions between Natural Regions (Parkland and Grassland) in the Southern inventory of equivalent disturbance levels.

		Disturbance (%)				
Type	**Metric**	**0-20**	**20-40**	**40-60**	**60-80**	**80-100**
Shape	SHAPE_AM	0.46	0.452	0.104	0.727	0.488
	CIRCLE_AM	0.847	0.976	**0.013**	0.614	0.72
Aggregation	AI	0.445	0.407	0.124	0.806	0.502
	COHESION	0.183	0.538	0.107	0.824	0.637
	CONTAG	0.951	0.452	0.194	0.225	0.135
	ENN	0.393	0.282	0.706	0.512	0.886
	SPLIT	0.612	0.97	**<0.001**	0.598	0.515
Diversity	SIDI_wet	0.188	0.994	**<0.001**	0.142	0.677
	SIDI_land	0.558	0.654	0.205	0.683	0.718

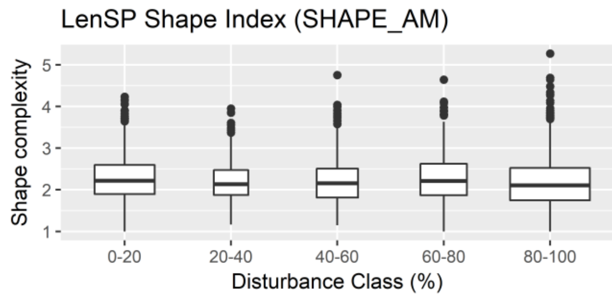
Appendix 10: Results of the pairwise comparisons for the Southern Parkland

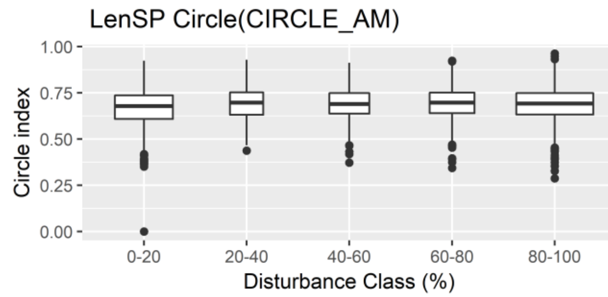
Disturbance class (%)	**0-20** n=22 CI=20.89	**20-40** n=17 CI=23.77	**40-60** n=14 CI=26.19	**60-80** n=16 CI=5.5
20-40 n=17 CI=23.77	SIDI_land***			
40-60 n=14 CI=26.19	SIDI_land***	SIDI_land**		
60-80 n=16 CI=24.5	SIDI_land***	SIDI_land***	SIDI_land***	
80-99.99 n=72 CI=11.55	SIDI_land***	SIDI_land***	SIDI_land***	SIDI_land***

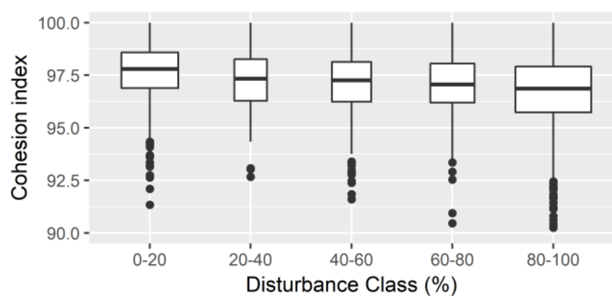
* p=<0.05; **p<0.01; ***p<0.001

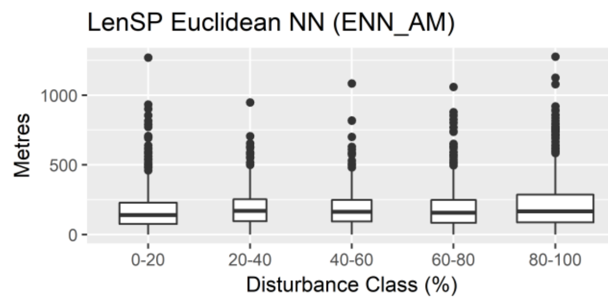
Note that the wide range of confidence intervals associated with each comparison.

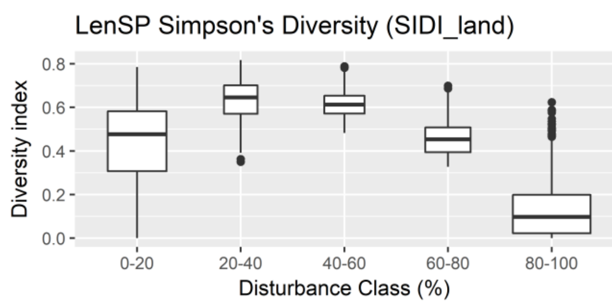
Appendix 11: Boxplots of LenSP Central inventory

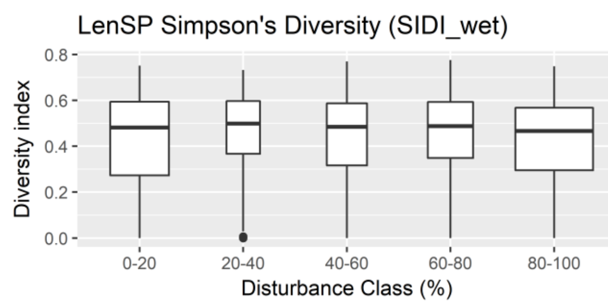


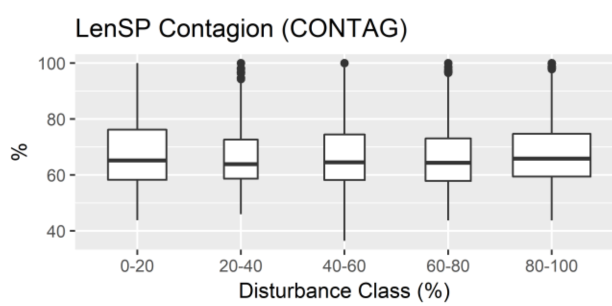


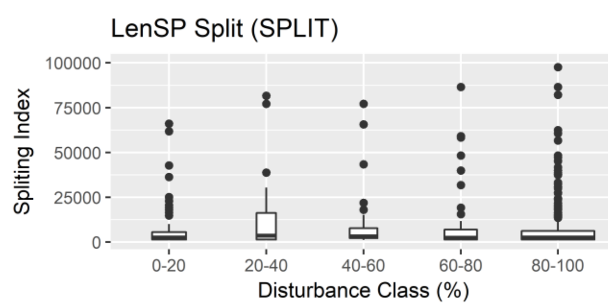


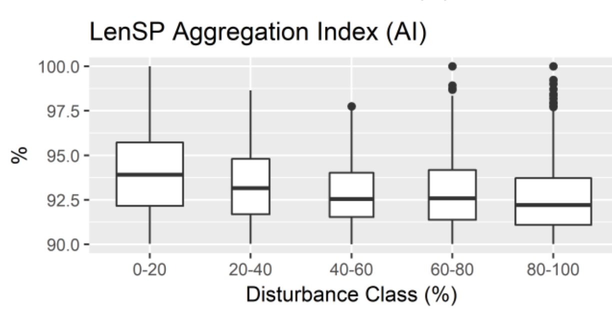










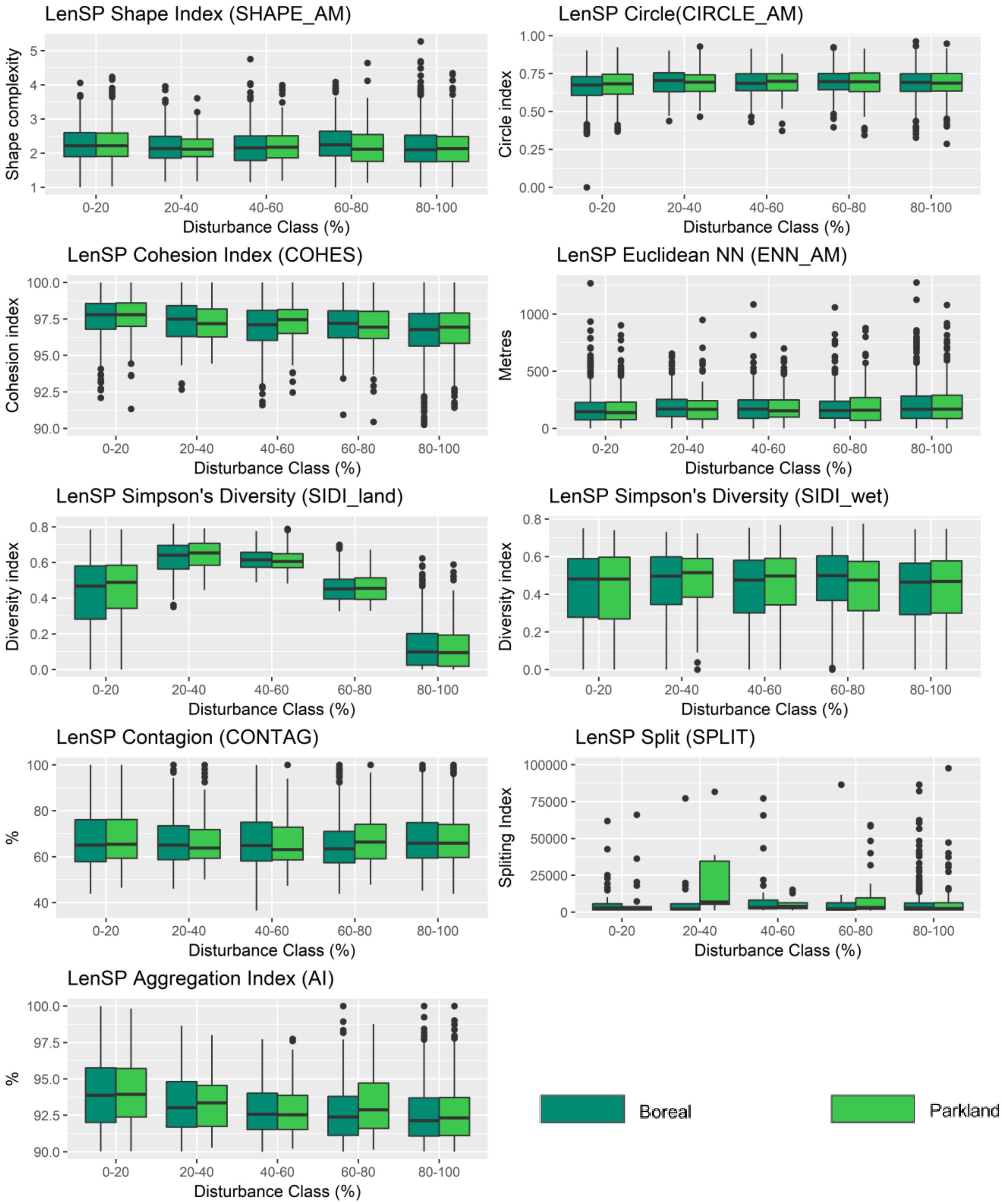


Appendix 12: LenSP pairwise comparison for the Central inventory

Disturbance Class (%)	0-20	20-40	40-60	60-80
20-40	ENN_MN* COHESION** SIDI_land*** SPLIT***			
40-60	AI** COHESION*** SIDI_land*** SPLIT*** SIDI_wet*** ENN_MN*	SIDI_land*** SPLIT*		
60-80	CIRCLE_AM* COHESION*** SPLIT*** SIDI_land*** ENN_MN* AI***	SIDI_land*** SPLIT*** COHESION*	SIDI_land*** SPLIT*	
80-99.9	SHAPE_AM*** CIRCLE_AM* COHESION*** ENN_MN*** SIDI_land*** SPLIT*** SIDI_wet*	COHESION*** SIDI_land*** SIDI_wet* SPLIT*** AI*** ENN**	COHESION*** SIDI_land*** SPLIT*** AI*** SIDI_wet** CONTAG**	SHAPE_AM** COHESION** SIDI_land*** AI*** SPLIT***

* p=<0.05; **p<0.01; ***p<0.001

Appendix 13: Boxplots of LenSP Central inventory separated by natural region



Appendix 14: LenSP pairwise of Central inventory Boreal region

Disturbance Class (%)	0-20	20-40	40-60	60-80
20-40	COHESION*** ENN_MN* SIDI_land*** SPLIT**			
40-60	AI*** COHESION*** ENN_MN* SIDI_land*** SPLIT***	AI** SIDI_land***		
60-80	CIRCLE_AM* COHESION*** SIDI_wet* CONTAG* SPLIT*** AI*** ENN_MN** SIDI_land***	SHAPE_AM* COHESION* SIDI_land*** SPLIT** AI***	SHAPE_AM* SIDI_land***	
80-99.9	SHAPE_AM** CIRCLE_AM* COHESION*** ENN_MN*** SIDI_land*** SIDI_wet* SPLIT*** AI***	COHESION*** SIDI_land*** SIDI_wet* SPLIT*** AI***	COHESION** CONTAG** SIDI_land*** SPLIT*** AI*** SIDI_wet	SHAPE_AM*** AI*** COHESION** SIDI_land*** SPLIT*** CONTAG**

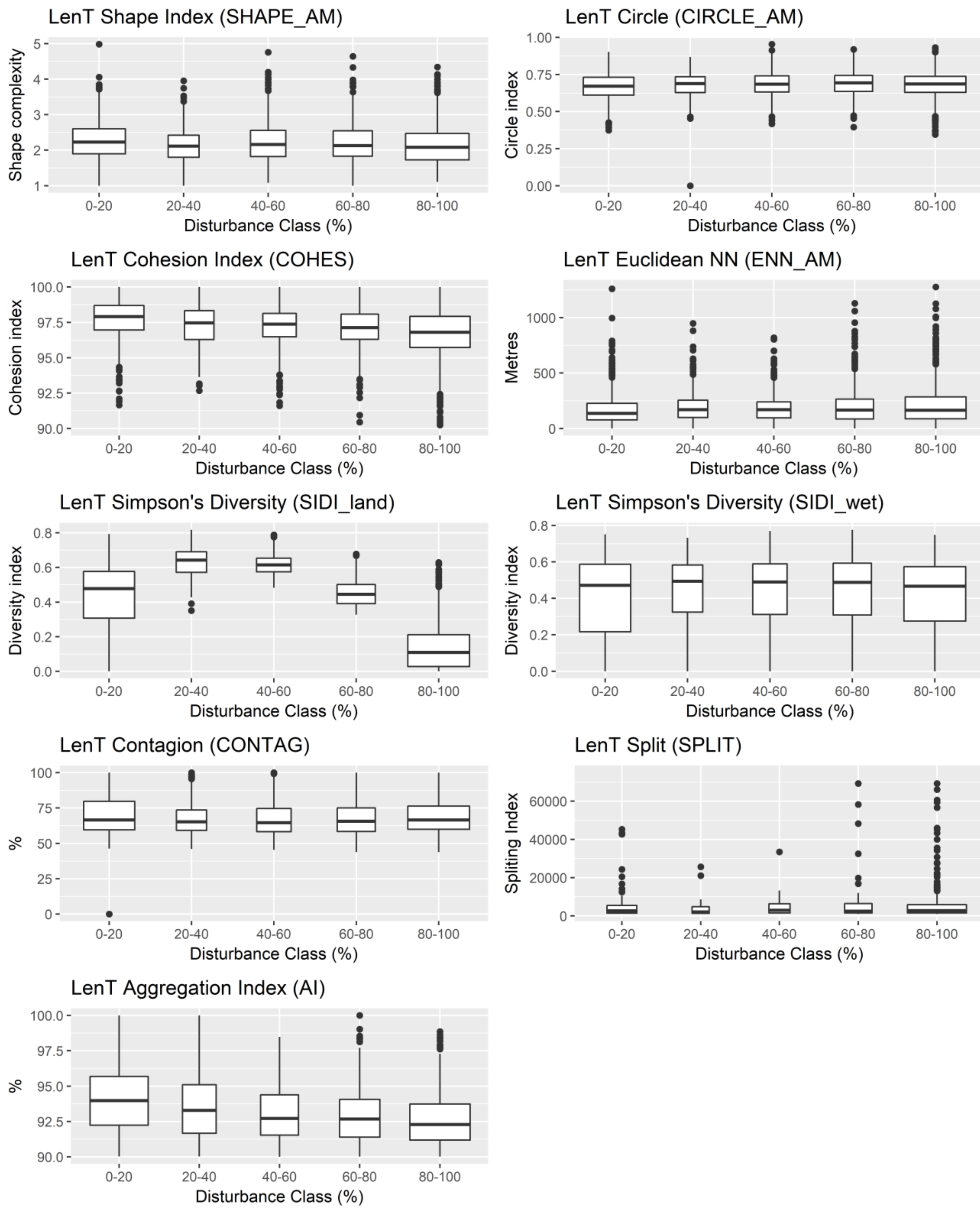
* p=<0.05; **p<0.01; ***p<0.001

Appendix 15: LenSP pairwise of Central inventory Parkland region

Disturbance Class (%)	0-20	20-40	40-60	60-80
20-40	COHESION*** ENN_MN* SIDI_land*** SIDI_wet* SPLIT*** AI*			
40-60	COHESION*** ENN_MN* SIDI_land*** SPLIT*** AI*** CIRCLE*	AI** SPLIT* SIDI_land		
60-80	CIRCLE_AM* COHESION*** SPLIT*** AI*** ENN_MN** SIDI_land***	COHESION* SIDI_land*** SPLIT*** AI***	SIDI_land***	
80-99.9	SHAPE_AM*** CIRCLE_AM* COHESION*** ENN_MN*** SIDI_land*** SIDI_wet*** SPLIT*** AI***	COHESION*** SIDI_land*** SIDI_wet*** CONTAG* SPLIT*** AI***	COHESION*** SIDI_land*** SIDI_wet* CONTAG* SPLIT*** AI***	SHAPE_AM*** COHESION** SIDI_land*** SIDI_wet*** CONTAG** SPLIT*** AI***

* p=<0.05; **p<0.01; ***p<0.001

Appendix 16: Boxplot plots for LenT for the Central inventory

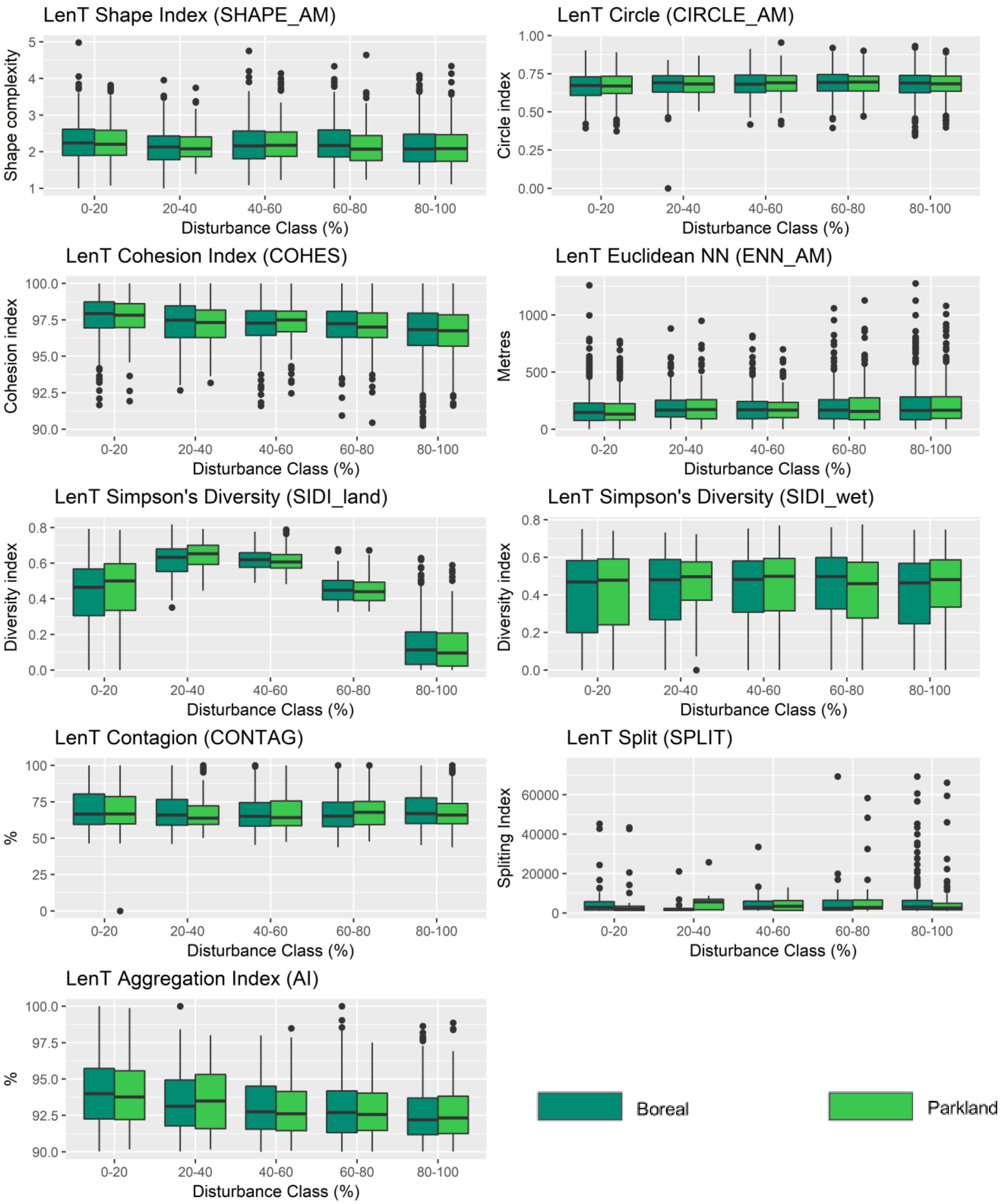


Appendix 17: LenT pairwise comparison of Central inventory

Disturbance Class (%)	0-20	20-40	40-60	60-80
20-40	SHAPE_AM* COHESION** ENN_MN* SIDI_land*** SPLIT*** AI*			
40-60	COHESION*** AI*** ENN_MN* SIDI_land*** SPLIT***	SIDI_land***		
60-80	CIRCLE_AM* COHESION*** ENN_MN** SPLIT*** AI*** SIDI_land***	SIDI_land*** SPLIT** AI*** COHESION*	SIDI_land*** SPLIT**	
80-99.9	SHAPE_AM*** CIRCLE_AM* COHESION*** ENN_MN*** SIDI_land*** SPLIT*** AI*** SIDI_wet*	COHESION*** SIDI_land*** SPLIT*** AI*** SIDI_wet*	SHAPE_AM* COHESION*** SIDI_land*** SPLIT*** AI*** SIDI_wet** CONTAG**	COHESION** SIDI_land*** SPLIT*** AI***

* p=<0.05; **p<0.01; ***p<0.001

Appendix 18: Boxplots of LenT Central inventory separated by natural region



Appendix 19: LenT pairwise comparison of Central inventory Boreal region

Disturbance Class (%)	0-20	20-40	40-60	60-80
20-40	SHAPE_AM** AI* COHESION** ENN_MN* SIDI_land*** SPLIT***			
40-60	SHAPE_AM* AI*** COHESION*** ENN_MN* SIDI_land*** CONTAG* SPLIT***	AI* SIDI_land***		
60-80	CIRCLE_AM* COHESION*** ENN_MN** AI*** SIDI_land*** SPLIT***	SIDI_land*** SPLIT* AI*** COHESION*	SIDI_land***	
80-99.9	SHAPE_AM*** CIRCLE_AM* COHESION*** ENN_MN*** SIDI_land*** SPLIT*** SIDI_wet AI***	COHESION*** SIDI_land*** SPLIT*** AI*** SIDI_wet*	SHAPE_AM* COHESION*** SIDI_land*** SIDI_wet* CONTAG** SPLIT*** AI***	SHAPE_AM** COHESION*** SIDI_land*** SIDI_wet AI*** SPLIT***

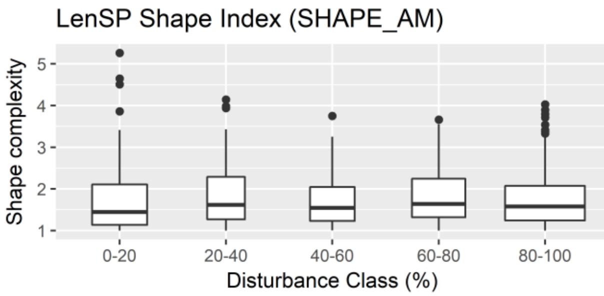
* p=<0.05; **p<0.01; ***p<0.001

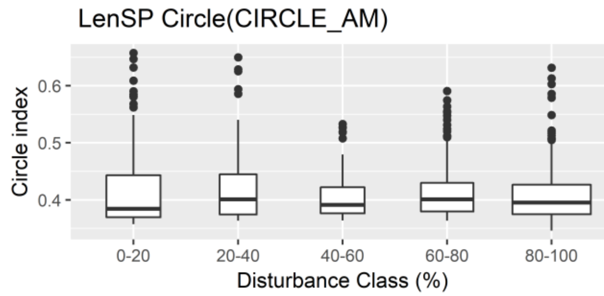
Appendix: 20 LenT pairwise comparison of Central inventory Parkland region

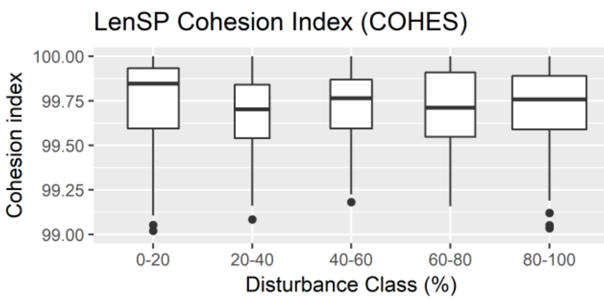
Disturbance Class (%)	0-20	20-40	40-60	60-80
20-40	COHESION** ENN_MN* SIDI_land*** SPLIT*** AI*			
40-60	COHESION*** ENN_MN* SIDI_land*** SPLIT*** AI**	SIDI_land***		
60-80	SHAPE_AM** CIRCLE_AM* COHESION*** ENN_MN** SPLIT*** AI*** SIDI_land***	SIDI_land*** SPLIT AI* COHESION*	SHAPE_AM* AI* SIDI_land*** SPLIT***	
80-99.9	SHAPE_AM** COHESION*** ENN_MN*** SIDI_land*** SPLIT*** AI*** CIRCLE_AM*	COHESION*** SIDI_land*** SPLIT*** AI***	SHAPE_AM* COHESION*** SIDI_land*** SPLIT*** AI***	COHESION** SIDI_land*** AI*

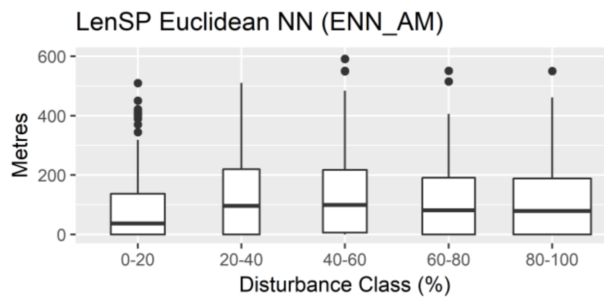
* p=<0.05; **p<0.01; ***p<0.001

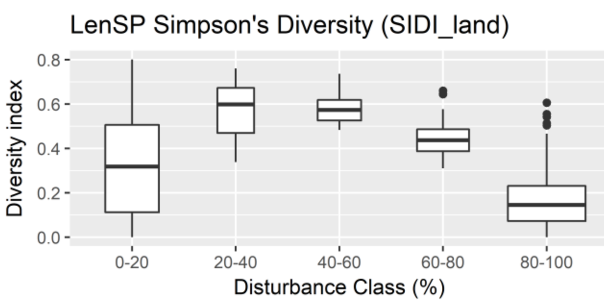
Appendix 21: Boxplots of LenSP Southern inventory

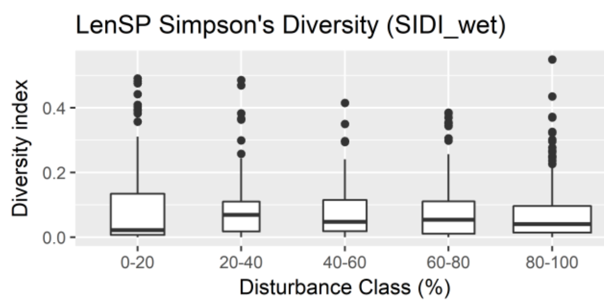


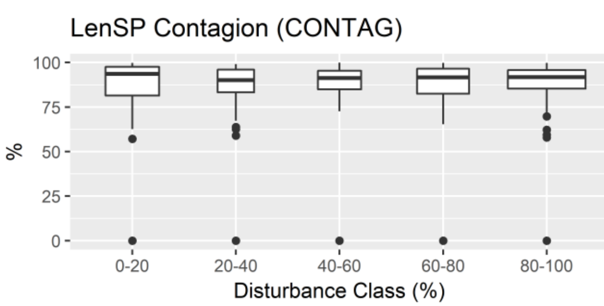


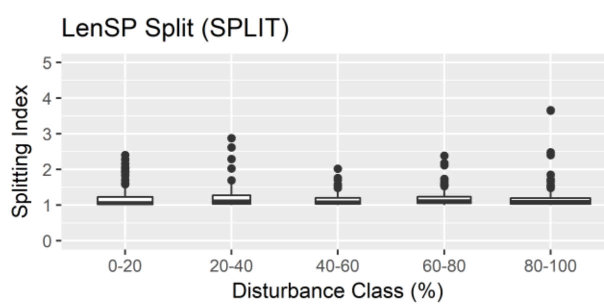


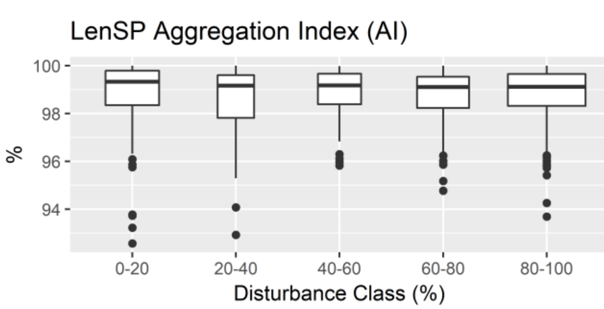










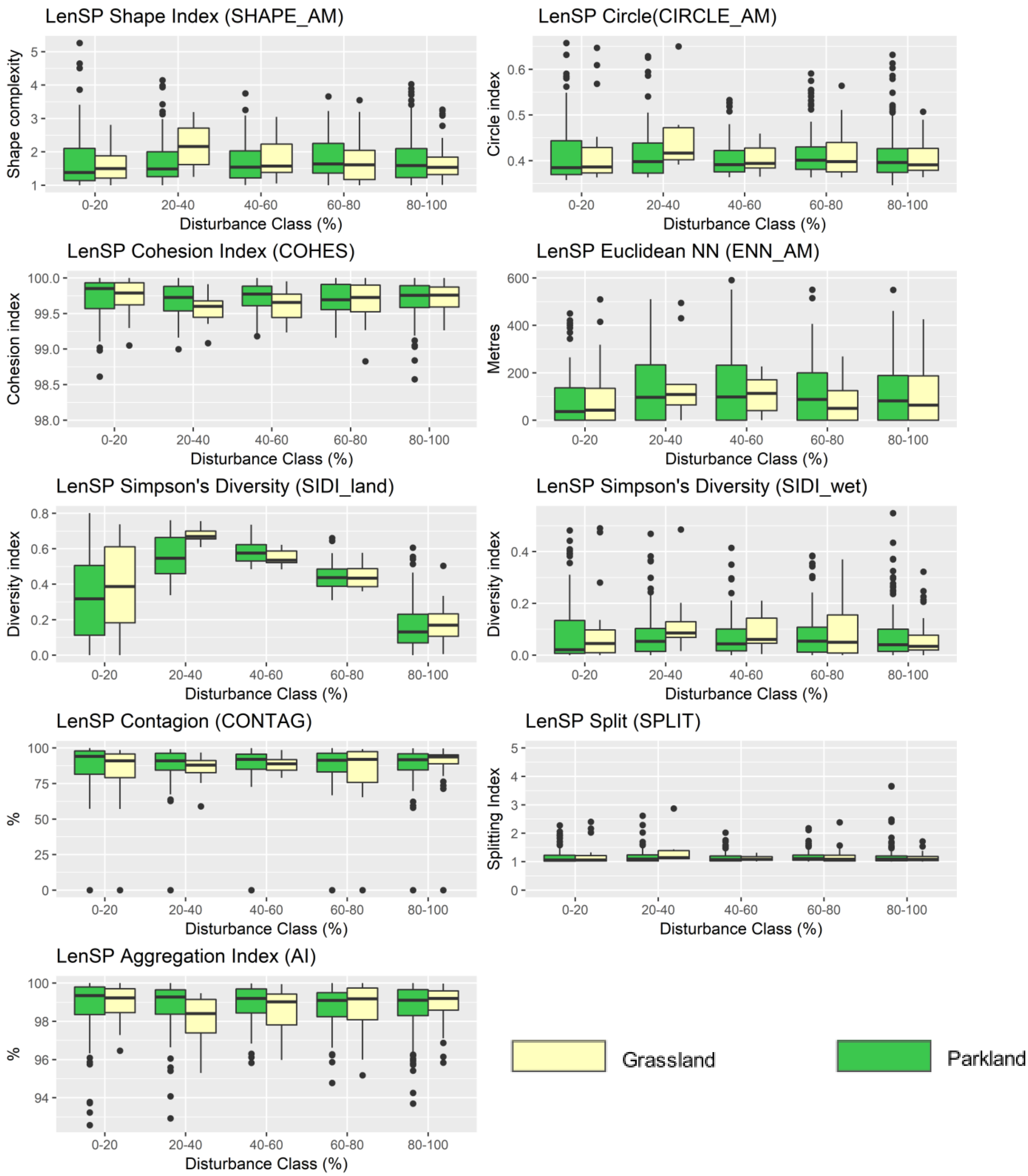


Appendix 22: LenSP pairwise comparison for Southern inventory

Disturbance Class (%)	0-20	20-40	40-60	60-80
20-40	ENN* SIDI_land***			
40-60	ENN* SIDI_land***	SIDI_land***		
60-80	SIDI_land***	SIDI_land***	SIDI_land***	
80-99.9	CIRCLE_AM* ENN** SIDI_land***	SIDI_land***	SIDI_land***	SIDI_land***

* p=<0.05; **p<0.01; ***p<0.001

Appendix 23: Boxplots of LenSP Southern inventory separated by natural region



Appendix 24: LenSP pairwise comparison of the Southern inventory Parkland region

Disturbance Class (%)	0-20	20-40	40-60	60-80
20-40	SHAPE_AM* CIRCLE_AM* COHESION* SIDI_land*** SIDI_wet* SPLIT* AI*			
40-60	SIDI_land***	SPLIT* SIDI_land**		
60-80	SIDI_land***	SHAPE_AM* CIRCLE* SIDI_land*** SIDI_wet* SPLIT* AI*	SIDI_land***	
80-99.9	SIDI_land***	SHAPE_AM* CIRCLE_AM* COHESION* SIDI_land*** SIDI_wet* CONTAG* SPLIT** AI*	SIDI_land***	SIDI_land***

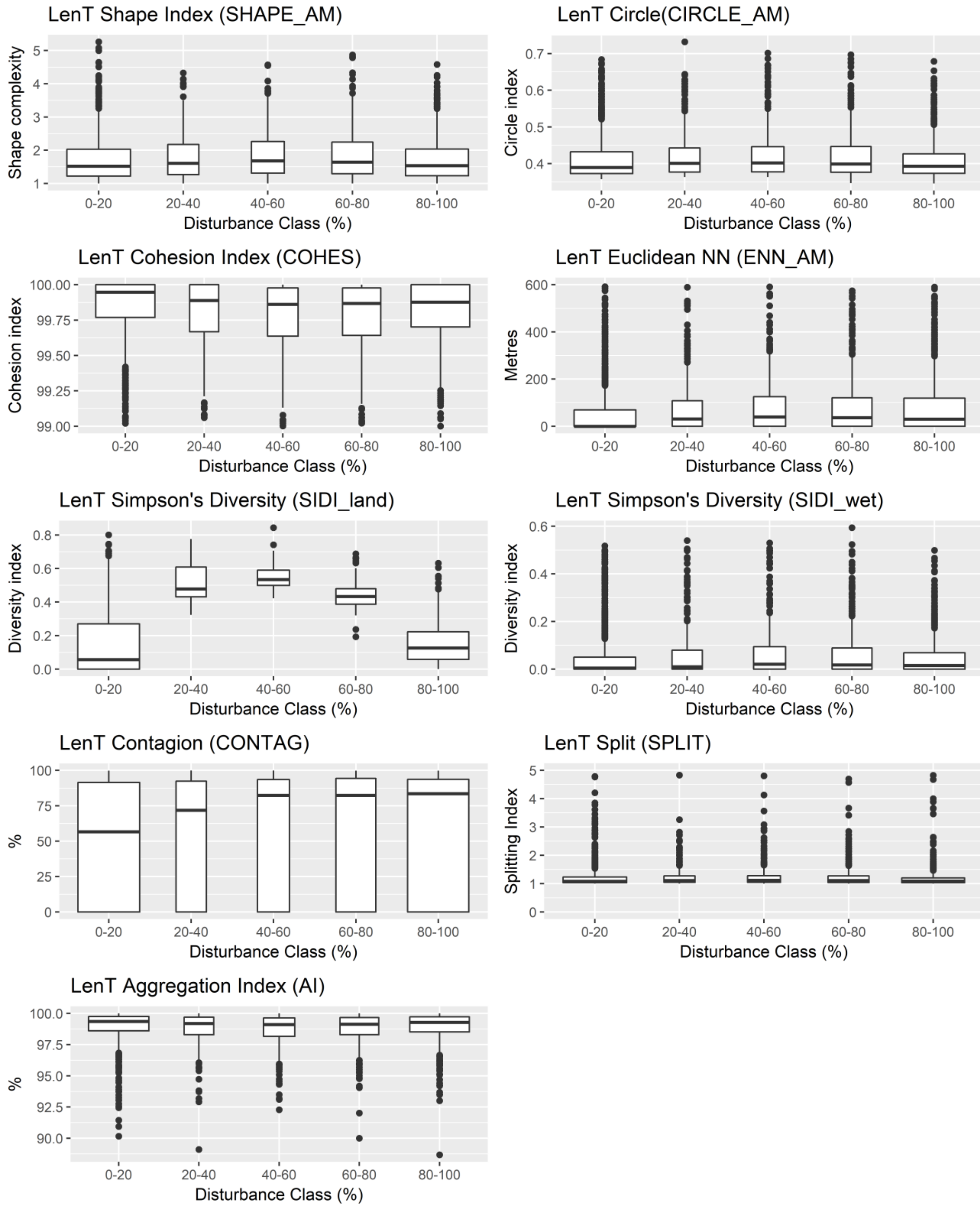
* p=<0.05; **p<0.01; ***p<0.001

Appendix 25: LenSP pairwise comparison of the Southern inventory Grassland region

Disturbance Class (%)	0-20	20-40	40-60	60-80
20-40	ENN** SIDI_land***			
40-60	ENN** SIDI_land***	SIDI_land***		
60-80	SHAPE_AM* CIRCLE* ENN** SIDI_land*** CONTAG* SPLIT* AI*	SIDI_land***	SIDI_land***	
80-99.9	ENN*** SIDI_land*** CONTAG*	SIDI_land***	SIDI_land***	SIDI_land***

* p=<0.05; **p<0.01; ***p<0.001

Appendix 26: Boxplots of LenT Southern inventory

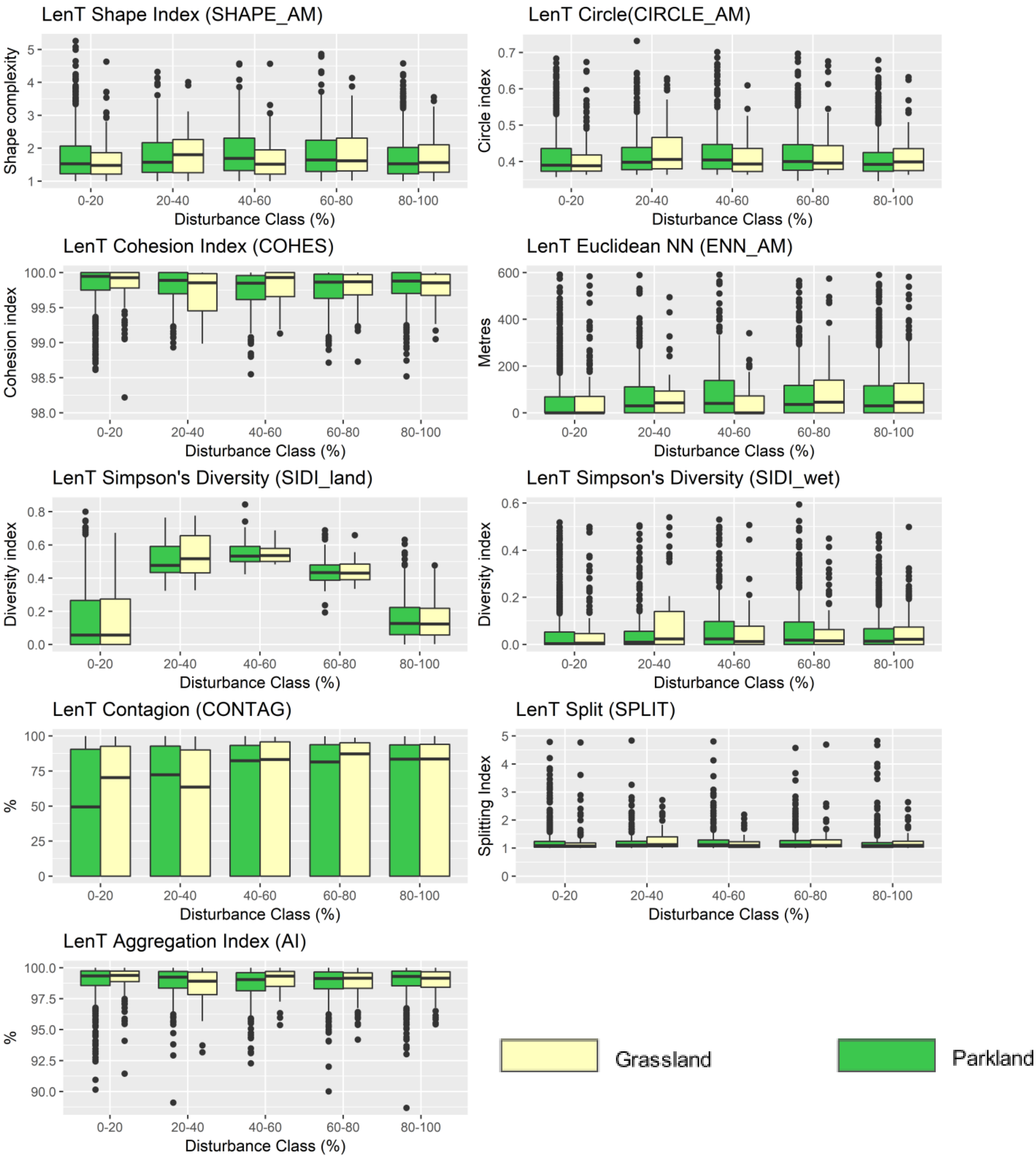


Appendix 27: LenT pairwise comparison of Southern inventory

Disturbance Class (%)	0-20	20-40	40-60	60-80
20-40	SIDI_land***			
40-60	SHAPE_AM* AI* SIDI_land*** CONTAG*** SPLIT* AI*	SIDI_land***		
60-80	SHAPE_AM* AI*** COHESION*** SPLIT** SIDI_land*** CONTAG**	SIDI_land*** CONTAG* COHESION***	SIDI_land*** COHESION***	
80-99.9	COHESION*** SPLIT** SIDI_land*** CONTAG** CIRCLE_AM**	CIRCLE_AM* SIDI_land*** CONTAG** SPLIT* COHESION***	AI* CIRCLE_AM** SIDI_land*** SPLIT* COHESION***	SHAPE_AM* CIRCLE_AM** SIDI_land*** SPLIT** COHESION** AI*

* p=<0.05; **p<0.01; ***p<0.001

Appendix 28: Boxplots for LenT Southern inventory separated by natural region



Appendix 29: LenT pairwise comparison on the Southern inventory Parkland region

Disturbance Class (%)	0-20	20-40	40-60	60-80
20-40	SHAPE_AM** SIDI_land*** SIDI_wet**			
40-60	SIDI_land*** AI* SIDI_wet*	SIDI_land*** SIDI_wet**		
60-80	SHAPE_AM** AI** COHESION* SIDI_land*** SIDI_wet* CONTAG**	SIDI_land*** CONTAG* SIDI_wet*	ENN* SIDI_land***	
80-99.9	SHAPE_AM** CIRCLE_AM* SIDI_land*** SPLIT** SIDI_wet* CONTAG**	SIDI_land*** CONTAG**	COHESION* SIDI_land*** AI* SPLIT*	SIDI_land*** AI* CIRCLE_AM** COHESION* SIDI_wet* SPLIT**

* p=<0.05; **p<0.01; ***p<0.001

Appendix 30: LenT pairwise comparison of Southern inventory Grassland region

Disturbance Class (%)	0-20	20-40	40-60	60-80
20-40	SIDI_land*** SHAPE_AM***			
40-60	AI*** SIDI_land*** SIDI_wet* SHAPE_AM** COHESION*	SIDI_land*** SIDI_wet** SHAPE_AM* AI*		
60-80	SIDI_land*** SIDI_wet* CONTAG** SIDI_land*** AI***	SIDI_land*** SIDI_wet* CONTAG*	SIDI_land***	
80-99.9	CONTAG** SPLIT**	CONTAG** SIDI_land***	SHAPE_AM*** AI*** COHESION* ENN* SIDI_land*** SPLIT*	SHAPE_AM*** CIRCLE_AM** COHESION* SIDI_land*** SIDI_wet* SPLIT**

* p=<0.05; **p<0.01; ***p<0.001