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An Automated Framework to Identify Lost and Restorable Wetlands in the Prairie Pothole Region

Ann Waz, *The University of Western Ontario*

Supervisor: Dr. Irena Creed, *The University of Western Ontario*

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Abstract

While progress has been made in automating wetland identification, identifying lost and restorable wetlands remains a challenge. A suite of automated methods was developed and applied to the Nose Creek watershed near Calgary, Alberta to establish a historical wetland inventory and the proportion of permanently versus temporarily lost wetlands. A power-law function of wetland area vs. wetland frequency using wetlands derived from the fusion of a high resolution digital elevation model and near-infrared data identified permanent loss of 11.0% by number and 0.6% by area. The difference between historical and existing wetlands was used to estimate a further temporary loss of 61.1% by number and 78.3% by area. Historical wetlands lost to ditch drainage are easily restored by ditch plugging. Therefore, an algorithm was created using digital terrain analysis that distinguished drainage ditches intersecting wetlands using surface curvature. The 1,588 ditch-drained wetlands identified represent a potential recovery of 11.7% of the temporary loss by number and 12.5% by area. Automated techniques to estimate wetland loss and identify priority wetlands for restoration provide powerful tools for wetland management.

Keywords: Prairie pothole region, wetland, management, drainage ditches, restoration

Co-Authorship Statement

A manuscript submitted as a result of this work will include Ann Waz as the first author as she developed the method, completed data processing, and wrote the thesis. Dr. Irena F. Creed will be co-author on any manuscript for her contributions to refining the method, interpreting the results, editing the manuscript, and providing financial support. This Master's thesis was funded by a Natural Sciences and Engineering Research Council (NSERC) Canada Graduate Scholarship Masters (CGS-M) awarded to Ann Waz, and an NSERC Discovery Grant and Alberta Land Institute Research Grant awarded to Irena F. Creed.

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List of Abbreviations

CWI	Canadian Wetland Inventory
DEM	Digital Elevation Model
DUC	Ducks Unlimited Canada
GIS	Geographic Information Systems
LiDAR	Light Detection and Ranging
MMU	Minimum Mapping Unit
NWI	National Wetland Inventory
P	Precipitation
p_{dep}	Probability of Depression
PET	Potential Evapotranspiration
PPR	Prairie Pothole Region
SPOT	Système pour l'observation de la Terre

Chapter 1

1 Introduction

1.1 Problem Statement

Wetland management, including protection and restoration of wetlands, begins with understanding where wetlands are located. While progress has been made in the geographic information systems (GIS) and remote sensing fields to automate wetland identification (Lang et al. 2012; Tiner et al. 2015; Serran and Creed 2016), the automated identification of lost and restorable wetlands still faces challenges (Dahl and Watmough 2007; Clare and Creed 2014). Within the Prairie Pothole Region, prairie potholes have largely been lost to agricultural activity including filling and draining (Dahl 2014; Watmough and Schmoll 2007). In face of their continued loss, wetland policies are increasingly favouring protection and restoration to maintain and re-establish valued wetland ecosystem functions. Wetland inventories, which include information on lost and restorable wetlands, serve as vital components of wetland management strategies, providing a scope of wetland loss and informing priorities for where wetlands should be protected and restored. Simple, automated techniques are needed to support these wetland management decisions.

1.2 Scientific Justification

1.2.1 The Prairie Pothole Region and its Wetlands

The Prairie Pothole Region (PPR) covers 777,000 km^2, extending across central North America, from Alberta, Saskatchewan and Manitoba in the north, toward Montana, North and South Dakota, Minnesota and Iowa in the south (Dahl 2014). The retreat of the Wisconsin glacier left behind millions of depressional wetlands in the fine-grained glacial till, known as prairie potholes (Johnson et al. 2008). The dry seasonal climate varies across the PPR, becoming wetter towards the east, and warmer towards the south (Johnson et al. 2005). In addition to precipitation and temperature gradients, the PPR is also prone to cycles of drought and deluge (Winter and Rosenberry 1998). The semiarid climate supports grasslands as the dominant ecosystem, much of which been converted to agricultural activity which is currently the largest land use (Gleason et al. 2008; Environment Canada 2013).

Prairie potholes are characterized as wetlands surrounded by uplands (Tiner 2003). They tend to be small (<1 ha) (van der Valk and Pederson 2003; Watmough and Schmoll 2007), and shallow (< 1 m in depth) (Huang et al. 2011). Natural surface drainage networks have not fully developed among these depressional wetlands (Winter 1989), resulting in wetlands exhibiting a continuum of connectivity – from geographically isolated to permanently connected to other waters bodies (USEPA 2015). Prairie potholes depend on snowmelt and precipitation as sources of water (Winter 1989), and the presence of surface water varies greatly, with temporary, seasonal, and semi-permanent wetlands being the most common wetland types (Stewart and Kantrud 1971; Kantrud et al. 1989). Concentric circles of vegetation are typical, including deep marsh vegetation, shallow marsh vegetation, wet meadow vegetation, and low prairie vegetation, reflecting the varying moisture gradients within wetlands (Kantrud et al. 1989). Many wetlands usually dry out by the fall (Smith et al. 1964; Dahl 2014).

Prairie potholes are valued components of the landscape, with their diverse functions often dependent on the degree of hydrologic permanence and connectivity. The ability of these wetlands to store runoff during precipitation events can decrease peak flows, and thereby reduce flooding potential (Tiner 2003). By serving as locales for groundwater recharge, prairie potholes aid in the stabilization of water supplies (McLaughlin et al. 2014). They serve as sinks for nutrients, either by sequestration in the sediments or by transformation to gaseous forms, thereby enhancing water quality (Marton et al. 2015). Prairie potholes also serve as important habitat for waterfowl and as hotspots for endemism and biodiversity (Leibowitz 2003).

1.2.2 Wetland Loss

Unfortunately, up to 70% of wetlands in the PPR have been lost (Dahl and Watmough 2007), largely due to agriculture (Dahl 2014; Watmough and Schmoll 2007). The Canadian Prairie Habitat Joint Venture monitoring program revealed that between 1985 and 2001, 6% of wetland basins were lost, with 62% of the lost area replaced by cultivation, 21% replaced by perennial grass cover, and 6% replaced by infrastructure and development (Watmough and Schmoll 2007). Farmers alter wetlands to increase property access, increase cultivated area and subsequently crop yield, and increase the diversity of options of crops that can be planted (Van der Gulik et al. 2000; Blann et al. 2009). Ephemeral, temporary and seasonal wetlands are most vulnerable to human alteration to enhance agricultural activities (Stewart and Kantrud 1973; Reynolds et al. 2006; Bartzen et al. 2010).

Mechanisms of anthropogenic wetland alteration include ditch drainage, subsurface drainage, cultivation, and filling (Figure 1.1). Human made drainage ditches facilitate surface drainage and are one of the most common mechanisms of wetland alteration (Government of Manitoba 1985; Watmough and Schmoll 2007; Blann et al. 2009). A drainage ditch is dug to carry water away from a wetland and can also be dug along natural drainage patterns, such as those formed from fill and spill. Drainage ditches vary morphologically, but are usually between 1 and 10 m wide and up to 1 m deep. The water can be carried to a variety of locations including roadside ditches, creeks, dugouts, as well as larger wetlands.

Subsurface drainage, also known as tile drainage, uses a network of underground perforated pipes to divert water. Subsurface drainage can be used to target only a wet area, or a large network can be used to control the water table under entire agricultural fields, termed pattern tile drainage (Euliss et al. 2014). Subsurface drainage is not common within the Canadian Prairies, likely due to their higher cost compared to surface ditches as well as their potential to become blocked by ice, particularly in Alberta during winter Chinooks (Government of Manitoba 1985; Watmough and Schmoll 2007). The use of tile drains increases in the southern portion of the PPR (Dahl 2014).

Some wetlands can also be directly cultivated without drainage. Cultivation can be temporary in nature, such as only during dry years, or cultivation may occur only along the edges of wetlands. Filling and levelling wetlands is most commonly associated with infrastructure and urbanization, however, the repeated land levelling and sedimentation associated with cultivation can also result in the filling of wetland basins over time (Gleason and Euliss 1998; Watmough and Schmoll 2007).

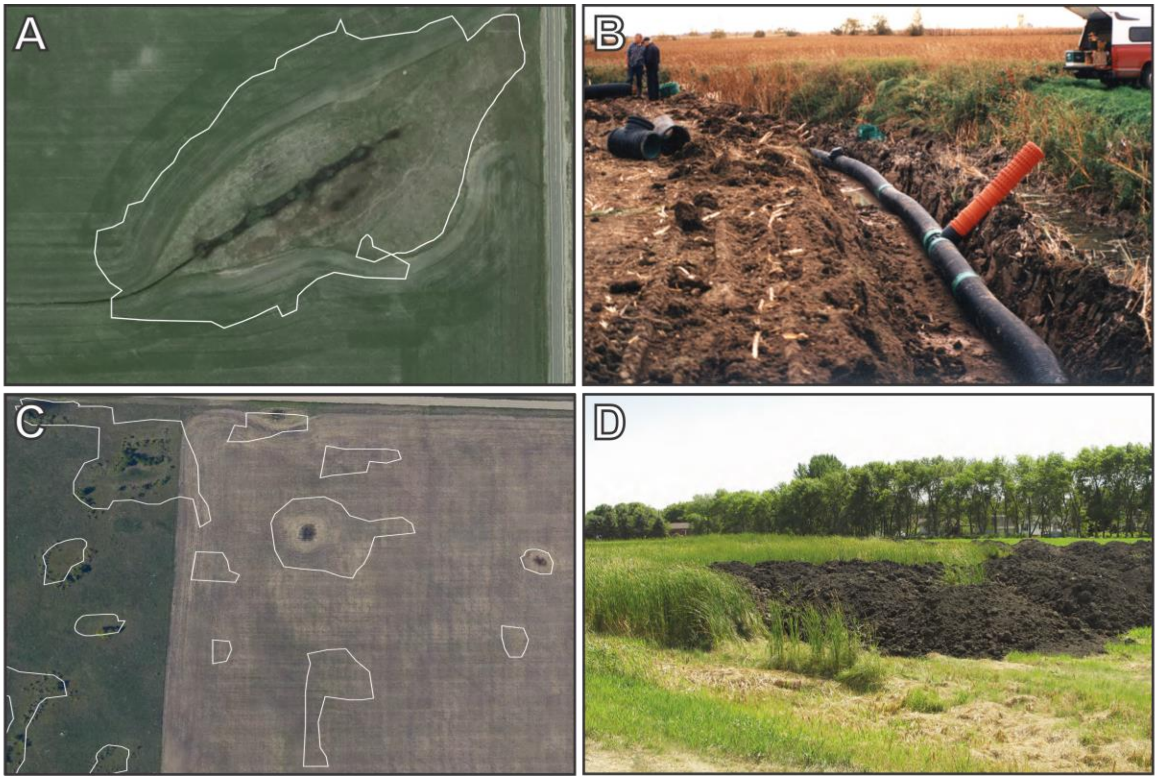

Figure 1.1 Wetlands (outlined in white) can be altered through: (A) ditch drainage; (B) subsurface drainage (Image retrieved from: Soleno n.d.); (C) cultivation, and (D) filling (Image retrieved from: Dahl 2014, p. 40).

1.2.3 Wetland Restoration

With the substantial loss of wetlands in the PPR, there has been growing interest in wetland restoration. Wetland restoration is a complex science and the effort required to restore a wetland will vary depending on factors such as hydrological regime, wetland size, and the duration and type of wetland impact (Weinhold and van der Valk 1989; Galatowitsch and van der Valk 1994). Each case of wetland restoration is unique, however, the wetlands that are easiest to restore are generally those that have minimal changes (Galatowitsch and van der Valk 1994). Cultivated wetlands with minimal filling and no artificial drainage are not usually considered to be permanently lost (Dahl 2014), since the wetland features are maintained and wetlands generally return once farming stops.

Drained wetlands have a high potential for restoration by eliminating human made drainage features to restore their hydrology. A wetland drained with a drainage ditch can be restored by filling a portion of the drainage ditch, known as ditch plugging (Figure 1.2) (Galatowitsch and van der Valk 1994). A tile drained wetland can be restored by breaking and removing portions of the perforated pipes (Galatowitsch and van der Valk 1994). In addition to restoring hydrology, the fragmented nature of wetlands within agricultural landscapes and the depletion of seed banks during cultivation means that the recovery of the wetland plant community often depends on artificial seeding (Weinhold and van der Valk 1989; Mulhouse and Galatowitsch 2003). Here, ditch-drained wetlands have an advantage for restoration because wetland plants can often grow in ditches, serving as seed banks for restored wetlands (Galatowitsch and van der Valk 1994).

While ditch-drained wetlands are considered a form of restorable wetland loss, filled wetlands are generally seen as a form of permanent wetland loss. Wetlands filled for the purpose of infrastructure and urbanization are unlikely to be restored. Wetlands filled due to land levelling and sedimentation would require the excavation of fill, re-contouring of the wetland depression, and revegetation in the restored wetland (Galatowitsch and van der Valk 1994). The high cost of restoring filled wetlands means that the restoration of filled wetlands is rarely pursued. Management strategies instead

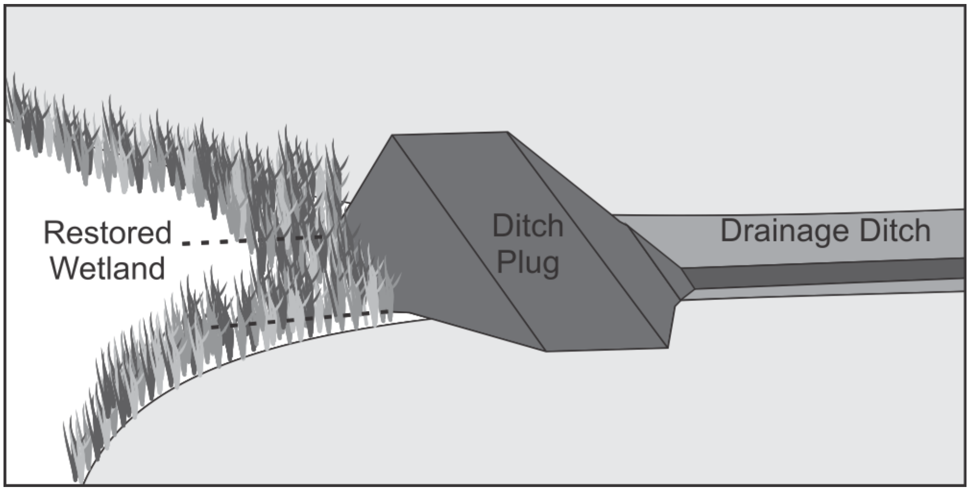

Figure 1.2 Ditch-drained wetlands can be restored by filling all or a portion of the drainage ditch, termed ditch plugging (Adapted from: Minnesota Board of Water and Soil Resources 2015, p. 3).

focus on the prevention of land levelling and sedimentation (Gleason and Euliss 1998), including the prioritization of restoration sites to where the potential for sedimentation is minimal (e.g. choosing to restore wetlands in pasture over wetlands in cropland) (Galatowitsch and van der Valk 1994).

1.2.4 State-of-science to Identify Existing, Restorable, and Lost Wetlands

Wetland inventories form the basis of wetland management. Several methods exist to identify existing, restorable, and lost wetlands. Methods to identify wetlands and delineate their boundaries have evolved from manual to increasingly automated methods (Lang et al. 2012; Tiner et al. 2015; Serran and Creed 2016), however, there is still room for improvement, especially with regard to restorable wetlands (Dahl and Watmough 2007; Clare and Creed 2014).

Manual methods to identify wetlands present on the landscape include the use of field surveys and aerial or satellite image interpretation. Wetland inventories for small areas have involved on-the ground surveys of wetlands, which while precise, are expensive and time consuming, even for small areas. Wetland inventories for larger areas have deployed professionals skilled in the interpretation of aerial photographs or satellite imagery to identify wetlands. While aerial photographs and satellite imagery are becoming increasingly available, the ability to easily update these inventories is a concern (Baker et al. 2007). For example, in the U.S., large portions of the National Wetland Inventory are based on imagery from the 1970s and 1980s (Tiner 2009). Manually derived inventories also use mapping resolutions that may miss small (<1 ha) wetlands. The Canadian Wetland Inventory (CWI) requires a minimum mapping unit (the smallest wetland that can be reliably mapped, MMU) of 1 ha or better (Fournier et al. 2007), which may not capture small (< 1 ha) prairie potholes.

Automated approaches to wetland mapping rely on digital image processing, including pixel-based classification and object-based image analysis. Automated mapping methods take advantage of a large variety of remotely sensed data including aerial photography, radar, multispectral imagery, and hyperspectral imagery, to automate wetland delineation (Ozesmi and Bauer 2002; Baker et al. 2006). Pixel-based classification techniques take advantage of the spectral signatures of different land use classes to sort pixels into different classes (Jensen 2005). For example, water and vegetation have unique spectral signatures in near-infrared (NIR) data. Classification techniques can be used to distinguish between wetland types (Dechka et al. 2002; Niemuth et al. 2010) as well as vegetation within wetlands (Phillips et al. 2005; Adam et al. 2010). Confusion among classes, which exists when classes have similar spectral signatures, can be improved with increased spatial and temporal data resolution, as well as ancillary data (Ozesmi and Bauer 2002).

The increased availability of fine resolution imagery (≤ 3m) has created opportunities to further improve automated wetland mapping techniques. In particular, object-based segmentation approaches to wetland classification are growing, classifying groups of pixels rather than individual pixels to better capture spatial context, thereby mimicking human interpretation (Dronova 2015; Knight et al. 2015). Serran and Creed (2016) applied object-based segmentation to a digital elevation model (DEM) derived from light detection and ranging (LiDAR) data to map wetlands based on topography, capturing even small and shallow prairie potholes. While the method successfully captured the diverse sizes of prairie potholes, some manual post-processing was still required to adjust wetland boundaries, especially in urban areas where flatter landscapes were mistaken for wetlands (Serran and Creed 2016).

Great strides have been made in automating wetland inventories and delineating even small wetlands such as prairie potholes. In contrast, the identification of lost and restorable wetlands has room to grow. At one end of the spectrum, probabilistic and statistical sampling programs, such as the Canadian Prairie Habitat Joint Venture program can monitor wetland change at regional and national scales (Dahl and Watmough 2007). The Canadian Prairie Habitat Joint Venture monitoring program conducts detailed sampling of 153 transects, stratified by sub-region, to understand wetland loss across the region (Watmough and Schmoll 2007). In the U.S.'s PPR, remotely sensed imagery and field verification for 755 random sample plots provides wetland status and change estimates for the region (Dahl 2014). These sampling programs may report on trends but neither generate the precision and accuracy of lost and restorable wetlands at the scales needed for effective wetland management. Specifically, they do not easily identify an inventory of potential candidates for restoration.

At the other end of the spectrum, spatially-based methods to identify wetland loss have used time series of wetland inventories, soil data, and well as DEMs. A historical time series of imagery can be used to create wetland inventories through time to identify historical extent and detect changes (Ozesmi and Bauer 2002). However, wetlands that are temporarily dry due to climate cycles may be misclassified as lost (Cowardin et al. 1981). Wetland change can also be inferred by analyzing hydric soil presence to determine the historical extents of wetlands (Galatowitsch and van der Valk 1994; Miller et al. 2009). Van Meter and Basu (2015) used the presence of depressions derived from a DEM, together with the presence of hydric soil to identify the historical extent of wetlands. The historical extent of wetlands was compared to the U.S. National Wetland Inventory (NWI) to identify wetland loss (Van Meter and Basu 2015).

Wetland loss can also be estimated using information from wetland area versus frequency plots. Due to the fractal nature of natural waterbodies, the data on these plots follow a power law (Downing et al. 2006; Zhang et al. 2009; Seekell et al. 2013; Van Meter and Basu 2015; Serran and Creed 2016), that is a negative linear relationship when plotted on logarithmic-logarithmic axes. Deviations from the power law have been used to provide non-spatial estimates of wetland loss (i.e., the difference between the power law trend and the deviation from the trend in existing wetlands provides an estimate of loss). When applied to topographically-based inventories (e.g., Serran and Creed 2016), deviations from the power law reveal estimates of permanent wetland loss - wetlands whose basins are no longer detectable on the landscape (i.e., filled wetlands). These are non-spatial estimates of permanent wetland loss, which is adequate given that these permanently lost wetlands are unlikely to be the focus of restoration efforts.

While wetland change detection methods quantify and identify the location of wetland loss, they do not answer whether a wetland is restorable. Methods which identify the mechanisms of wetland loss are required to assess whether a lost wetland is restorable. An improved ability to detect smaller features and the growing availability of high-resolution imagery and sensors now allows for the identification of restorable wetlands by targeting the mechanism of wetland loss. The ability to capture the mechanism of wetland loss varies with the footprint left by different wetland impacts. Drainage ditches leave visible changes on the surface, while surficial evidence of a tile drained wetland may consist only of an inlet pipe placed in the deepest portion of a basin (Biebighauser 2007). Therefore, the growing resolution of imagery holds promise for the identification of restorable, ditch-drained wetlands.

Drainage networks, both natural and human made, can be delineated using drainage algorithms (Schwanghart et al. 2013; Tarolli 2014), pixel classification (Liimatainen et al. 2015), object-based segmentation (Rapinel et al. 2015), or morphological filters (Bailly et al. 2008; Pirotti and Tarolli 2010; Passalacqua et al. 2012; Cazorzi et al. 2013). Previous methods used to map drainage networks, including the size of the study area, the location of the study area, the data and methods used, and the accuracy of the maps generated are summarized in Table 1.1. Drainage algorithms, while effective for natural streams, have been found to be ineffective at capturing human made drainage ditches, especially in flat areas (Schwanghart et al. 2013; Tarolli 2014). The success of pixel-based classification methods was found to be dependent on ditch depth (Liimatainen et al. 2015). Object-based segmentation methods have also been applied to LiDAR DEMs to identify ditch networks, with a minimum LiDAR point cloud data precision of 2 points per m^2 recommended (Rapinel et al. 2015). The increased availability of high resolution LiDAR data has also allowed for many studies to take advantage of morphological filters which use surface measurements to distinguish ditches (Bailly et al. 2008; Pirotti and Tarolli 2010; Passalacqua et al. 2012; Cazorzi et al. 2013). In particular, high resolution DEMs have been used to calculate and apply a threshold to the surface curvature to identify ditch networks (Pirotti and Tarolli 2010; Passalacqua et al. 2012). The progress that has been made in delineating drainage networks, when combined with the progress that has been made in automating the identification of wetland basins, can be brought together and applied to identify ditch-drained, restorable wetlands.

Table 1.1 Previous methods used to map drainage networks, including the size and location of the study area, data resolution and source, method used, and the accuracy of the maps generated.

Author	**Study Area**	**Resolution and Source**	**Method**	**Accuracy Results**
Schwanghart et al. 2013	15.5km² low relief, agricultural site (Midtjylland, Denmark)	1.6m LiDAR	Drainage algorithm	Algorithm failed to reconstruct the drainage network
Liimatainen et al. 2015	7.2km² peatland (Southern Ostrobothnia, Finland)	1.0m LiDAR	Supervised classification	F score of 0.98
Rapinel et al. 2015	2 wetland sites, 0.25km² each (Brittany, France)	4 points/m² LiDAR point cloud	Object-based image analysis	40.1% - 60.6% of reference network captured
Bailly et al. 2008	2.0 km² agricultural site (Languedoc, France)	10 points/m² LiDAR point cloud	Morphological filter: wavelet transform to detect concavities	70% overall accuracy, 50% omission rate, 15% commission rate
Pirotti and Tarolli 2010	0.7km² alpine basin (Eastern Italian Alps)	1.0m LiDAR	Morphological filter: curvature	Cohen's k of 0.488
Passalacqua et al. 2012	2880km² agricultural watershed (Minnesota, USA)	3.0m LiDAR	Morphological filter and drainage algorithm: curvature and upstream contributing area thresholds	N/A
Cazorzi et al. 2013	0.82km² agricultural site (North East Italy)	1.0m LiDAR	Morphological filter: residual topography (smoothed DEM – original DEM)	Captured 17km of the 19.5km network

1.3 Research Question and Thesis Objectives

This thesis focuses on building a comprehensive wetland inventory from which the following research question can be asked: What is the magnitude of existing, restorable (i.e., ditch-drained) and lost wetlands within an Alberta watershed?

To answer this research question, three objectives are specified. The first objective is to improve and fully automate an established wetland mapping method. The object-based segmentation method developed by Serran and Creed (2016) has proven to be particularly useful for prairie potholes, capturing even small and shallow wetlands. A limitation of this method is that it does not work in urban areas; rather, manual editing of the wetland inventory is required. Further automating wetland delineation in both agricultural and urban areas is important and therefore the method was advanced to take advantage of NIR imagery to delineate open water within urban areas, and thereby no longer necessitating the manual editing in urban areas. The refined wetland inventory is then used to create an inventory of historical wetlands.

The second objective of this thesis is to develop an automated method to identify restorable wetlands. Current methods of wetland loss detection do not identify mechanisms of wetland loss, which directly influences the restoration potential of a wetland. Ditch-drained wetlands are excellent candidates for restoration. To meet this objective, digital terrain analysis is used to identify drainage ditches and, together with the historical wetland inventory produced in Objective 1, ditch-drained wetlands are identified.

The third objective of this thesis is to bring wetland inventories together to estimate the number and area of existing, restorable, and lost wetlands. The historical wetland inventory (Objective 1), restorable wetland inventory (Objective 2), and the existing CWI will be compared to understand the extent of wetland loss and the potential for wetland restoration. A better understanding of the magnitude of wetlands loss as well as the spatial distribution of restorable wetlands would aid in the restoration of not only the number, area but also the distribution of wetlands within a watershed, a common goal in wetland restoration efforts.

1.4 Thesis Organization

This thesis follows a monograph format. Chapter 1 introduces the state of wetland loss within the Prairie Pothole Region and the need for automated tools to build comprehensive wetland inventories that include both historical inventories, contemporary inventories, and the proportion that are permanently or temporarily lost. Chapter 2 describes the test area, the Nose Creek watershed, including its climate, hydrology and ecology. Chapter 3 details the method developed to identify historical wetlands, existing wetlands, restorable wetlands, and estimate wetland loss. Chapter 4 presents the results of applying the methods to the Nose Creek watershed. Chapter 5 discusses the strengths and limitations of the developed methods in the context of the literature. Finally, Chapter 6 presents the main research findings and future research directions. The appendix includes supplemental details on the method.

Chapter 2

2 Test Area

2.1 Nose Creek Watershed

The Nose Creek watershed (51°16’57”N, 114°7’14”W) is located along the northern edge of Calgary, Alberta, covering 886 km^2 within the Prairie Pothole Region (Figure 2.1). Nose Creek and its tributary West Nose Creek join before entering the Bow River, which subsequently drains to the South Saskatchewan River and ultimately Lake Winnipeg. The watershed is characterized by a dry continental climate with a mean annual temperature of 4.4°C and a mean annual precipitation of 418.8 mm/yr, according to Canadian Climate Normals for 1981 - 2010 (Environment Canada 2015) (Figure 2.2). A negative water balance is characteristic of the area, with large amounts of potential evapotranspiration (calculated using the Hamon method, (Hamon 1961)) exceeding precipitation, resulting in a mean annual moisture deficit of -97.6 mm based on the period from 1948 to 2014 (Figure 2.3).

Past glaciation has deposited fine-grained glacial drift over the area resulting in dense silty or clayey tills (Winter 1989). The watershed is dominated by Black Chenozemic soil (Agriculture and Agri-Food Canada 2016) underlain by the Paskapoo Formation which comprises sandstones, mudstones and siltstones (Hamblin 2004). The topography ranges from 1336 m to 1048 m above sea level. The landscape consists largely of rolling and undulating plains, with wetlands, specifically marshes and seasonal ponds, located within depressions in this low-permeable terrain (Natural Regions Committee 2006). Prairie pothole wetlands are fed by snowmelt and spring rains and generally become dry through summer and fall (Winter 1989).

The watershed lies within the Parkland and Grassland natural regions (Natural Regions Committee 2006). These natural areas are largely cultivated with low amounts of native vegetation remaining. The Parkland natural region in the western half of the watershed is characterized by aspen forests and willow scrublands mixed with grasslands. Balsam poplar and white spruce can also be found on moist sites. The Grassland natural

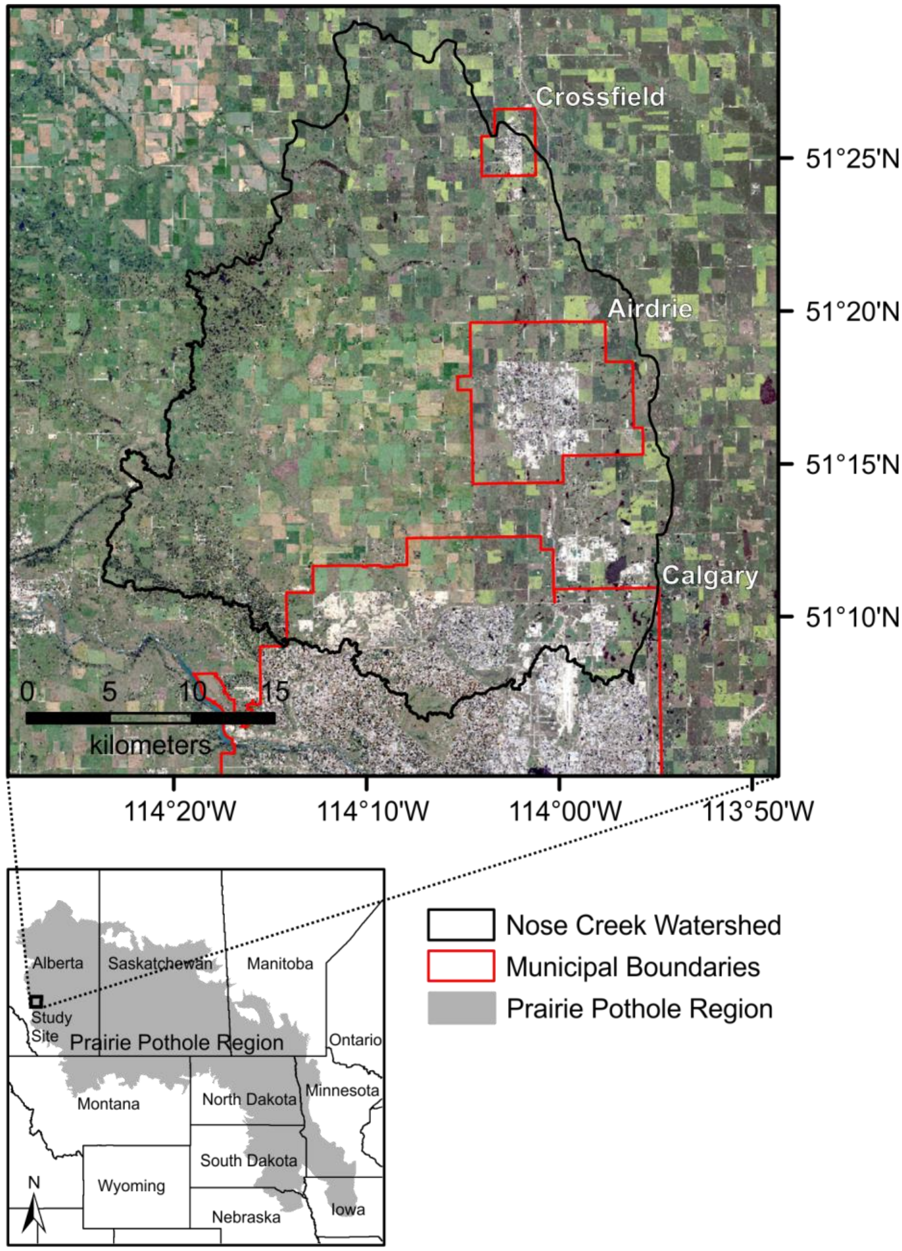

Figure 2.1 Map showing the location of the Nose Creek watershed, Alberta, Canada. The watershed is largely dominated by agricultural activities.

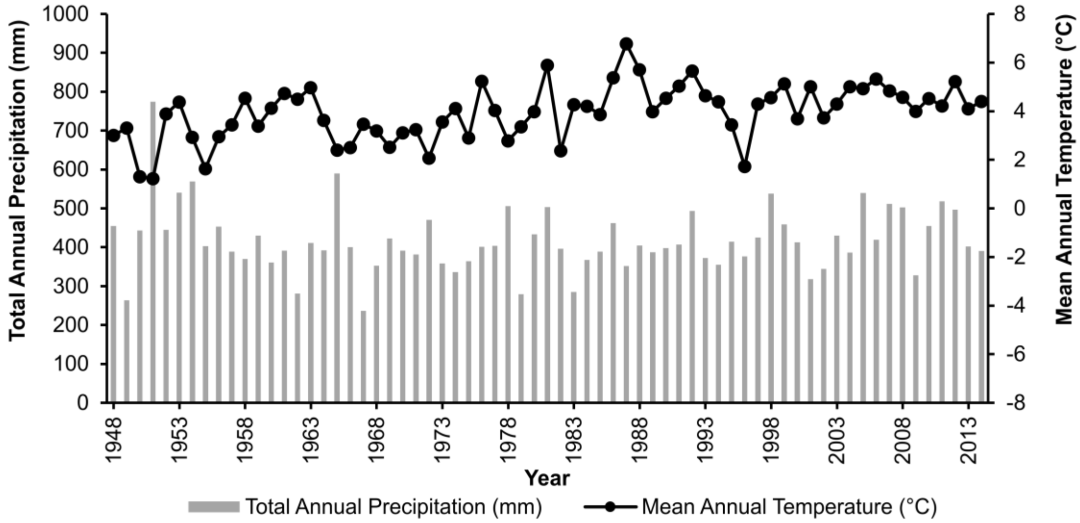

Figure 2.2 Total annual precipitation (mm) and mean annual temperature (°C) from 1948 to 2014 for the Calgary International Airport.

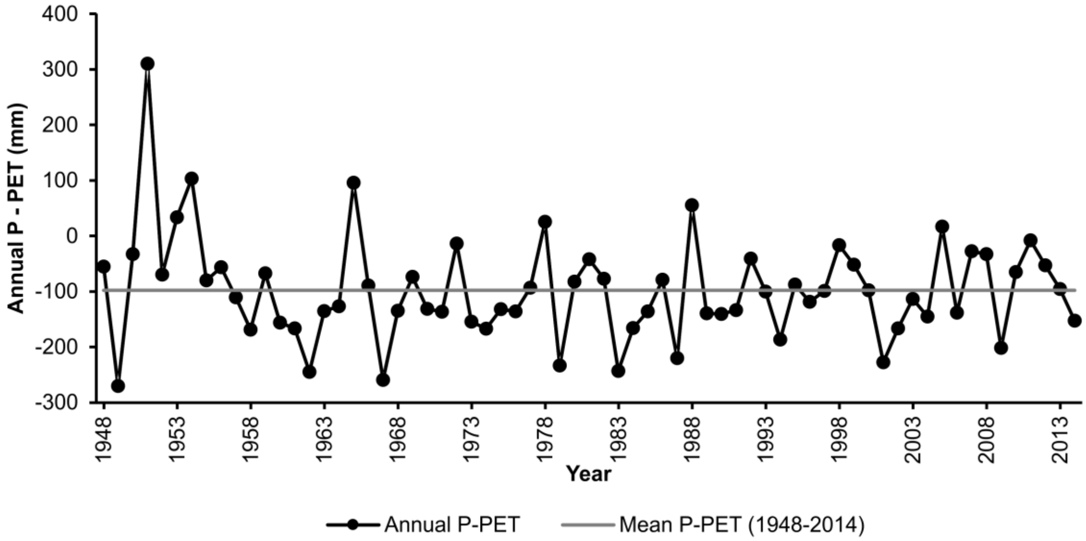

Figure 2.3 Time series of annual precipitation (P) minus potential evapotranspiration (PET) from 1948 to 2014 for the Calgary International Airport, with PET estimated using the Hamon (1961) method. The mean P-PET for the time period is presented by the dashed grey line.

region in the eastern half of the watershed is characterized by grasses. In ungrazed or moderately grazed sites mountain rough fescue, creeping juniper, Parry oat grass, bluebunch fescue and June grass can be found (Natural Regions Committee 2006).

Agricultural activities occur over approximately 70% of the watershed (Agriculture and Agri-Food Canada 2013). The most common agricultural crops include canola, spring wheat, barley and alfalfa (Government of Alberta 2012). Where the terrain is not favourable to crops, grazing predominates. The dominance of agricultural activity in the watershed is appropriate for the development of a method to identify wetlands altered due to agricultural activity, namely wetland drainage through surface ditches.

Chapter 3

3 Methods

An inventory of potential wetlands (i.e., including temporarily lost: with the wetland depression remaining; and existing wetlands) was delineated by adapting a previously developed technique that maps wetland depressions on the landscape (Serran and Creed 2016). Using the power law relationship between area vs. frequency of the potential wetlands, an estimate of historical wetlands was obtained, including those that have been permanently lost, as estimated by deviations from the power law distribution, and temporarily lost, as estimated by differences between the historical wetlands and existing wetlands. Restorable wetlands, specifically ditch-drained wetlands, were identified by developing a method to identify drainage ditches in the potential wetlands using digital terrain analysis of a DEM. The data layers used in this study, including their resolution, minimum resolvable unit, year of collection, and source are listed in Table 3.1.

3.1 Identifying Potential Wetlands

A flow chart of the method used to delineate potential wetlands is shown in Figure 3.1 and Figure 3.2. The method consists of four main steps that will be described in each of the following four sections, including stochastic analysis, object-based segmentation, near-infrared segmentation, and the consolidation of a final potential wetland inventory.

3.1.1 Stochastic Analysis

A LiDAR DEM with a horizontal resolution of 1 m and a vertical accuracy of 15 cm formed the basis of the mapping of potential wetlands. The raw LiDAR point cloud data, which had an average point density of 5.5 points per square metre, were pre-processed by Airborne Imaging Inc. which triangulated the data to form a triangular irregular network (TIN) before converting it to a 1 m raster. The LiDAR data were captured between October 14 and 17, 2014 during leaf-off period. LiDAR cannot penetrate water, therefore, the LiDAR acquisition dates were designed to fall when canopy was not present and during the driest part of the year to capture the underlying

Table 3.1 List of data layers used in this study including their resolution, minimum mapping unit (MMU) or minimum resolvable unit (MRU), time of capture, and source. Where a MMU was not provided, it was calculated using the method by Tobler (1987).

Data Layer	Resolution	Minimum Mapping Unit / Minimum Resolvable Unit	Source Data Year(s)	Creator/Source
Canadian Wetland Inventory	0.25 m	MMU = 0.02 ha	2006	Ducks Unlimited
DEM	1 m	MMU = 0.0009 ha	October 14, 17, 2014	Airborne Imaging Inc.
Roads	1:20,000	MRU = 0.04 ha	2016	AltaLIS Ltd.
Rail	1:20,000	MRU = 0.04 ha	2016	AltaLIS Ltd.
Hydrography	1:20,000	MRU = 0.04 ha	2016	AltaLIS Ltd.
Crop Map	30 m	MMU = 0.81ha	2013	Agriculture and Agri-Food Canada

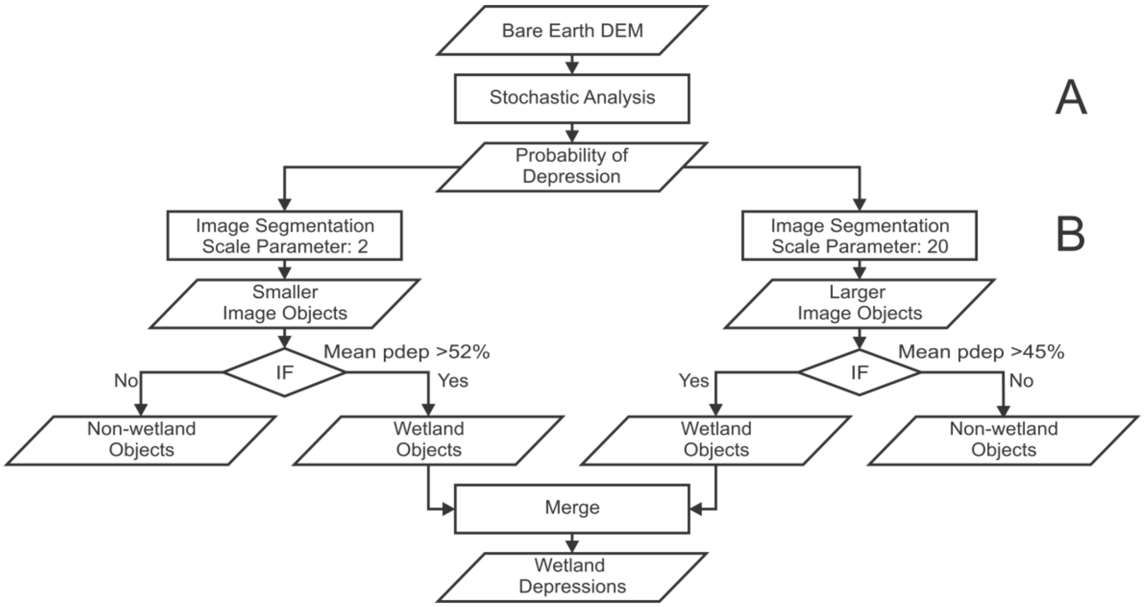

Figure 3.1 Flow chart of steps to delineate wetland depressions, including (A) stochastic analysis and (B) object-based segmentation. The flow chart continues in Figure 3.2 with steps (C) and (D).

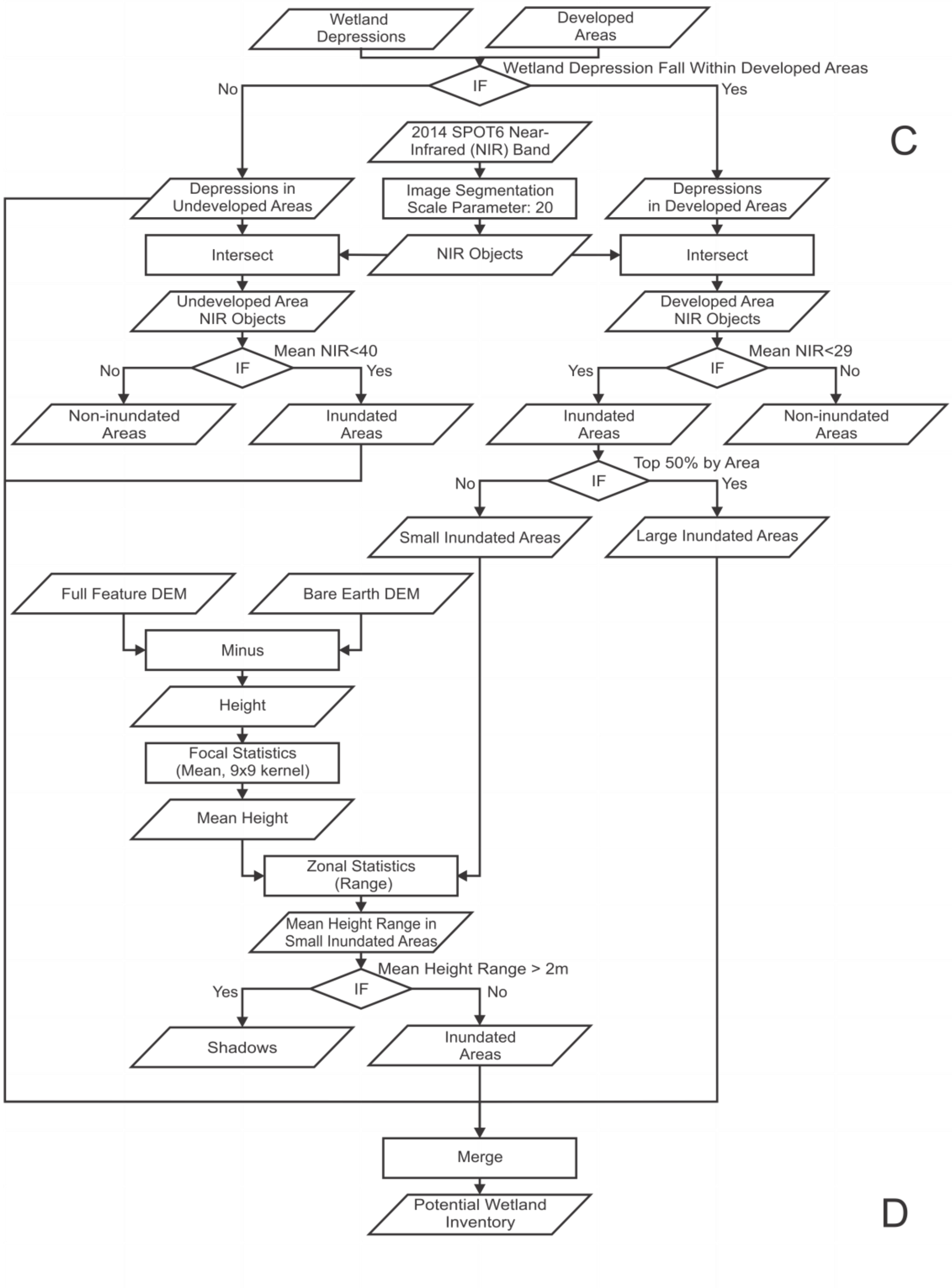

Figure 3.2 Flow chart of steps to delineate wetland depressions, continued from Figure 3.1, includes (C) object-based segmentation of near-infrared imagery to produce (D) a final inventory of wetland depressions and inundated areas.

topography of seasonally inundated areas. Both a bare earth and full feature DEM were provided.

The 1 m bare earth DEM was resampled using bilinear interpolation to a 3 m resolution, a resolution that was optimal for potential wetland mapping (i.e., the 1 m resolution produced artifacts; data not shown). Potential wetland depressions were identified by low lying topographic depressions surrounded by uplands. Digital terrain analysis in the form of stochastic modelling was used to identify the probability of a depression (p_{dep}) (Lindsay and Creed 2005). Using a Monte Carlo simulation, a random error term selected from the standard deviation of the distribution of random error terms equal to the 15 cm vertical accuracy of the DEM was added to the DEM. The Planchon and Darboux (2001) depression filling algorithm was then applied to the error-added DEM, and those pixels that were filled were flagged as depressions. This process of adding a random error to the DEM and subsequently applying a depression filling algorithm was iterated 1,000 times, and the final output was the probability of occurance of a depression, calculated as the proportion of times each pixel was identified as a depression. This probabilistic approach takes into account uncertainty from DEM error and distinguishes true depressions on the landscape as opposed to artifacts in the data (Lindsay and Creed 2006). Similar stochastic analyses have been used to identify wetlands in a variety of landscapes, including landscapes covered with forests and shallow (< 2 m in depth) soils (Creed et al. 2003), as well as landscapes covered with sparse forests and much deeper soils (Serran and Creed 2016). Stochastic analysis was performed using the Terrain Analysis System version 2.0.9 software (Lindsay 2005).

3.1.2 Object-based Segmentation

The p_{dep} map underwent object-based segmentation and classification. Object-based segmentation grouped similar pixels to create objects, which were then classified based on user-defined rules. Object-based segmentation has been successfully used in wetland mapping (Serran and Creed 2016), and is found to better detect the complexity of natural wetland boundaries compared to pixel-based approaches (Dronova 2015). Definiens eCognition Developer software (Trimble Navigation Limited 2009) was used for object-based segmentation.

During segmentation, pixels were merged into objects to minimize heterogeneity within the object. Segmentation occurred at two scales, termed multi-resolution segmentation, using scale parameters of 2 and 20. The scale parameters served as the heterogeneity thresholds, with objects halting growth when the scale parameter was surpassed (Benz et al. 2004). A scale parameter which produces small objects, essentially pieces of wetlands, is commonly used as the generated objects can be later joined together (Dronova 2015). However, using only one small scale parameter can result in fragmented wetland objects (Serran 2014). Therefore, multi-resolution segmentation was used as it aids in capturing the complexity of wetland sizes across landscapes (Serran 2014; Knight et al. 2015).

The change in heterogeneity gauged by the scale parameter was calculated as a function of spectral heterogeneity and shape heterogeneity. *Spectral heterogeneity* refers to the heterogeneity of the input layer, in this case, the p_{dep} values. The change in spectral heterogeneity was calculated by comparing the standard deviation of the p_{dep} values within the objects before and after potential merging (Benz et al. 2004). *Shape heterogeneity* refers to the smoothness and compactness of an object's shape, which was calculated using the object's perimeter and area (Benz et al. 2004). The relative importance of spectral heterogeneity and shape heterogeneity in the calculation of the total change in heterogeneity were adjusted by assigning them different weights. Spectral heterogeneity was assigned 100% of the weight for heterogeneity calculations.

These segmentation parameters (the scale parameters and heterogeneity weights) were previously determined heuristically for a 3 m LiDAR DEM of the Beaverhill watershed located in the PPR, approximately 270 kilometers (km) north of the Nose Creek watershed (Serran and Creed 2016). In object-based image analyses, the most common method for parameter selection is through trial and error (Dronova 2015). Given that an objective method for choosing segmentation parameters has not yet been established (Dronova 2015), the segmentation parameters from this other related study were applied.

A road vector layer buffered 15 m on each side served as an additional input in segmentation to prevent wetland objects from crossing roads (i.e., if a wetland was intersected by a road, it was treated as two separate wetlands). A 15 m buffer was used because most roads (e.g., gravel roads, paved roads, multi-lane highways, etc.) fell entirely within this buffered area.

Following segmentation, the objects were classified as potential wetlands using a rule set. For smaller objects (scale parameter 2) to be classified as a wetland depression, the mean p_{dep} value within the object was 0.52 (52%) or greater, and the object could not fall within the road buffer. For larger objects (scale parameter 20), the mean p_{dep} value within the object was 0.45 (45%) or greater, and the object could not fall within the road buffer. These classification thresholds for the objects were selected based on previous work in the Beaverhill watershed, where the thresholds were calibrated to an established wetland inventory (Serran and Creed 2016).

3.1.3 Near-infrared Segmentation

Following the multi-resolution segmentation of the p_{dep} layer, the potential wetland inventory was improved by an object-based segmentation of NIR imagery to automatically delineate inundated areas within wetland depression boundaries (Figure 3.2, part C). This object-based segmentation was introduced to improve wetland delineation in areas where topography alone was not sufficient to capture wetland boundaries, namely very flat areas (i.e., slope < 2.5 degrees) and developed areas. In undeveloped areas, large flat areas including riparian zones and wetland complexes were sometimes included as one single object. In developed areas, which includes residential, commercial, and industrial areas, stretches of flat areas such as subdivisions and parking lots were included as depressions, areas which are not appropriate to include as wetland depressions.

Before proceeding with NIR segmentation, the potential wetlands were first checked and then cleaned. Following the multi-object (p_{dep}) segmentation, some potential wetlands included “tails” where drainage ditches existed. The clean tool from the ET Geowizards extension (Tchoukansi 2012) for ESRI ArcGIS served to simplify wetland depression boundaries, removing the tails (i.e., drainage ditches) which were not part of the potential wetland. The clean tool was only run on non-riparian wetland depressions to prevent narrow riparian features from being broken up or eliminated.

NIR segmentation was applied across the watershed to identify inundated areas within potential wetlands. The potential wetlands were segregated into two groups: those that fell within developed areas and those that did not. Developed area boundaries were manually delineated to include commercial, industrial, and high-density residential areas. The potential wetlands were segregated because in developed areas NIR boundaries served to replace p_{dep} boundaries, whereas in undeveloped areas inundated areas served as supplementary information to the topographically-based (p_{dep}) boundaries.

A pansharpened orthomosaic of the NIR band (760-890 nm) of a SPOT 6 satellite image consisting of four scenes collected between April 29 and July 9, 2014 with a resolution of 1.5 m was received from BlackBridge Geomatics Corp. NIR is appropriate for mapping inundated areas as water strongly absorbs in the NIR range, resulting in low reflectance values that distinguish these areas from other land cover and land use classes. The NIR imagery was captured in spring 2014, as leaf-off imagery is optimal for identifying inundated areas (Ozesmi and Bauer 2002). The NIR imagery was resampled to 3 m to align with the p_{dep} layer and ensure the created inundated area objects fall within the wetland depressions.

During object-based segmentation of the NIR data, a scale parameter of 20 was used. Two scale parameters were not necessary, as inundated areas were spectrally homogeneous resulting in similar objects regardless of the scale parameter. A larger scale parameter was favoured for computational efficiency. Given the varying shapes of inundated areas, there was no one desired shape that was sought, therefore the spectral heterogeneity was set to 100% influence in heterogeneity calculations.

Following NIR segmentation, the resulting objects were then classified using a NIR threshold. Single band thresholding models have been used successfully to distinguish between water and non-water objects (Frazier et al. 2003; Jain et al. 2005; Sass et al. 2007; Sun et al. 2012). An object was classified as inundated if it fell within a potential wetland and the object’s mean reflectance value fell under a specified NIR threshold. Two different NIR thresholds were used. For developed areas, objects were classified as inundated if they had a mean NIR reflectance value equal to or less than 29. For undeveloped areas, objects were classified as inundated if they had a mean NIR value equal to or less than 40. The NIR classification thresholds were determined using training data. Seventy-five training objects representing inundated areas were manually delineated in the developed and undeveloped areas. The NIR threshold represents the mean reflectance plus one standard deviation for the training objects in each area. The difference in NIR threshold between the two areas is likely due to differences is the depth and turbidity of the waterbodies that are present. Based on visual inspection within the watershed, open water bodies such as relatively deep and clear storm water ponds were more common in developed areas, whereas water bodies such as relatively shallow and turbid natural wetlands were more common in undeveloped areas.

In developed areas, additional post-processing of the inundated objects was required to remove shadows that had been misclassified as inundated areas. In the NIR range, shadows have a similar spectral response as water, commonly leading to misclassification. To distinguish shadows from true inundated areas, two criteria were used: the size of the object, with the assumption that shadows will be small; and the surrounding height, with the assumption that shadows are usually caused by surrounding buildings. In developed areas, the smallest 50% by area inundated objects were further analyzed for their surrounding height to filter out likely shadows.

Surrounding height was determined by subtracting the bare earth DEM from the full feature DEM creating a layer of the height above ground level. The focal statistics tool was used to calculate the mean height in a 9 × 9 pixel kernel. The zonal statistics tool was then used to extract the range of the heights in the inundated or shadowed objects. If an inundated object exists in a flat area, the surrounding changes in height will be minimal, while if an object exists near a tall building, the surrounding change in height will be greater. The appropriate height threshold to distinguish between inundated and shadowed objects was selected after testing a range of options (i.e., 0.5, 1.0 and 2.0 m). For example, using the threshold of 1.0 m, if the range of mean heights of an inundated object was greater than 1.0 m, the object was flagged as a shadow. The accuracy of the objects flagged as shadows was then individually assessed using SPOT 6 color imagery for 2014. Two metres was chosen as the threshold, which removed the large majority of shadows while minimizing the removal of true inundated areas. In a similar assessment in undeveloped areas, objects identified as inundated were rarely (< 0.5% of the time) found to be shadows, and therefore removal of these shadows using surrounding height would have resulted in a large removal of actual inundated objects.

For the final potential wetland inventory, the boundaries of wetlands were represented by the classified wetland object boundaries from p_{dep} segmentation in undeveloped areas, and the classified inundated area objects from NIR segmentation in developed areas. In developed areas, due to anthropogenic interference, the topographically-based classified wetland object boundaries from p_{dep} segmentation were not successful at identifying wetlands, and were therefore replaced by the classified inundated area objects. In undeveloped areas, the classified wetland object boundaries were appropriate, and classified inundated area objects were also mapped to serve as supplementary information when needed (e.g., large riparian areas). Any objects below 0.0081 ha were removed, an area equivalent to a 3 × 3 window of 3 m pixels, an estimate of the MMU.

3.1.4 Accuracy Assessment

To assess the accuracy of the potential wetland inventory, historical and contemporary imagery were evaluated to determine whether there was evidence that actual wetlands were present. Twenty-one different sources of aerial and satellite imagery from 16 different years were assessed, as summarized in Table 3.2. The accuracy of wetland boundaries can be very difficult to determine, as wetland size and shape varies with climate, therefore only the presence of a wetland at any point in time was considered.

Two discrete classes were considered: wetland and other. One hundred random potential wetland polygons and 100 random non-wetland polygons were assessed, for a total sample size of 200. The sample size follows general recommendations for 75-100 samples per class (Congalton and Green 2008). In a two-case scenario, such as wetland and non-wetland, binomial distributions can also be used to estimate sample size (Congalton and Green 2008). The sample size of 100 is also in line recommendations based on the binomial distribution, which for an expected accuracy of 85% and 5% allowable error, suggests a total sample size of 203 (Ginevan 1979). Random polygons

Table 3.2 List of aerial and satellite imagery used for accuracy assessment, listed in reverse chronological order.

Data Layer	Resolution (m)	Spectral bands (nm)	Acquisition Date	Creator / Source
2014 orthophotos	0.3 m	Color	July 2014	Rocky View County
2014 SPOT 6 Pansharpened orthomosaic imagery	1.5 m	Near-infrared: 760-890 nm Red: 625-695 nm Green: 530-590 nm Blue: 450-520 nm	April 29 – July 9 2014	BlackBridge Geomatics Corp.
2013 SPOT 6	1.5 m	Red: 625-695 nm Green: 530-590 nm Blue: 450-520 nm	July 31 - September 17, 2013	Government of Alberta
2012 orthophotos	0.3 m	Color	Spring 2012	Rocky View County
2012 SPOT 5	2.5 m	Panchromatic: 480-710 nm	August 28 – October 15, 2012	Government of Alberta
2011 SPOT 5	2.5 m	Panchromatic: 480-710 nm	July 24 – September 20, 2011	Government of Alberta
2010 orthophotos	0.3 m	Color	Fall 2009	Rocky View County
2010 SPOT 5	2.5 m	Panchromatic: 480-710 nm	April 17 – October 21, 2010	Government of Alberta
2009 SPOT 5	2.5 m	Panchromatic: 480-710 nm	May 24 - November 2, 2009	Government of Alberta
2008 SPOT 5	2.5 m	Panchromatic: 480-710 nm	Sept 28 – Oct 2 2008	Government of Alberta
2007 orthophotos	0.3 m	Color	Spring/Summer 2007	Rocky View County
2007 SPOT 5	2.5 m	Panchromatic: 480-710 nm	October 15, 2007	Government of Alberta
2006 SPOT 5	2.5 m	Panchromatic: 480-710 nm	May 24 – August 5, 2006	Government of Alberta
2005 orthophotos	0.5 m	Color	May 2005	Rocky View County
2005 SPOT 5	2.5 m	Panchromatic: 480-710 nm	May 30, 2005	Government of Alberta
2003 orthophotos	1.0 m	Panchromatic	October 2003	Rocky View County

2000 orthophotos	1.0 m	Panchromatic	2000	Rocky View County
1999 orthophotos	1.0 m	Color	1999-2001	Valtus Imagery Services Ltd.
1966 georectified aerial imagery	1:31,680 (0.64 m)	Panchromatic	August 1 – September 4, 1966	Alberta Environment and Parks
1962 georectified aerial imagery	1:31,680 (0.64 m)	Panchromatic	June 8 - September 25, 1962	Alberta Environment and Parks
1949-1951 orthophotos	1:63,360 (1.6 m)	Panchromatic	1949-1951	Alberta Biodiversity Monitoring Institute (ABMI)

were generated by generating random points within the watershed, which were then buffered to create circles with an average area equal to the average area of the 100 random wetland polygons, namely 0.50 ha. The random polygons were generated such that there was no overlap with any wetland polygons.

A polygon was classified as a wetland if it had any of the following features:

- A stream going through it (i.e., a riparian wetland)
- Inundated areas at any point in the historical imagery
- Wetland vegetation patterns (i.e., concentric bands of vegetation varying with moisture) at any point in the historical imagery

The classification accuracy was determined by generating a confusion matrix (error matrix) and then calculating the overall accuracy, producer's and user's accuracy, omission error, commission error, and Cohen's kappa. The overall accuracy was calculated as the total number of correctly classified polygons divided by the total sample number. The producer's accuracy was calculated for the two classes (wetland and non-wetland) by calculating the total number of correctly classified polygons divided by the total number of polygons in that class. The omission error (also referred to as the error of exclusion) was calculated as the producer's accuracy subtracted from one. The user's accuracy was calculated for the two classes (wetland and non-wetland) by calculating the total number of correctly classified polygons divided by the total number of polygons predicted to be in that class. The commission error (also referred to as the error of inclusion) was calculated as the user's accuracy subtracted from one. Cohen's kappa, a measure of overall accuracy which takes into account chance agreement, was also calculated (Congalton and Green 2008).

3.2 Identifying Historical Wetlands

A historical wetland inventory was created using an enhanced potential wetland inventory. The CWI was created by Ducks Unlimited Canada using stereo pairs of high resolution panchromatic aerial photographs from 2006 to capture existing wetlands on the landscape to a MMU of 0.02 ha (Ducks Unlimited Canada 2006) . Within the Nose Creek watershed, the CWI identifies 6,858 wetalnds with an area of 2938 ha. The potential wetland inventory fails to capture some of the wetlands in the CWI. Therefore, the CWI was joined to the potential wetland inventory, based on the assumption that if a wetland existed in the 2006 CWI, it should also be present in the potential wetland inventory (Clare and Creed 2014), creating the *enhanced* potential wetland inventroy. To allow for direct comparison with the CWI inventory, a MMU of 0.02ha was applied to the enhanced potential wetland inventory.

A piecewise linear regression was applied to the enhanced potential wetland data to identify the power law line and breakpoints where the data deviate from the power law line. The power law is based on the fractal nature of natural waterbodies (Downing et al. 2006), therefore, developed areas, where waterbodies are largely engineered, were removed from further analysis. Wetlands were binned by area, starting from the smallest wetland size of 0.02 ha. The bin increment was chosen objectively as the coarsest resolution of the data used to create the enhanced potential wetland inventory, which was 0.0009 ha (or 9 m^2), equivalent to the area of one grid on the 3 m LiDAR DEM. When applying a piecewise regression to the wetland area vs. wetland frequency data plotted on logarithmic-logarithmic scales, there are often two breakpoints, one breakpoint at a smaller wetland area, and a second breakpoint at a larger wetland area where the wetland frequency begins to be one. A three segment piecewise regression was first applied to identify these two breakpoints. Data with an area above the second breakpoint, and data with a frequency below the second breakpoint were removed (Serran and Creed 2016). A two segment piecewise regression was then run on the remaining data to define the power law line, the breakpoint, and the deviation of enhanced potential wetland data from the power law line.

The power law trend in the enhanced potential wetland inventory was then analyzed to estimate the historical wetland inventory. This power law relationship was extrapolated to the MMU (0.02 ha). If the observed enhanced wetland inventory data deviated from the power law line, this deviation was assumed to represent permanently lost wetlands (Serran and Creed 2016) (Figure 3.3). Given that the enhanced potential wetland inventory is largely topographically-based, wetland loss estimates derived from deviation from the power law line represented wetlands that are no longer

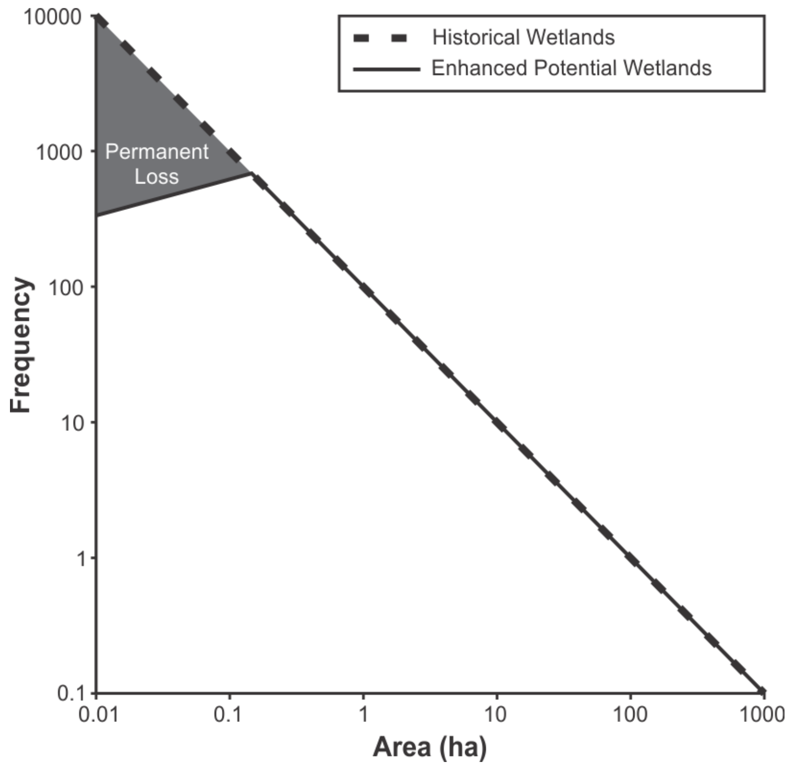

Figure 3.3 Conceptual plot of wetland area versus wetland frequency on logarithmic-logarithmic scales. Permanent loss is determined by calculating the number or area between the enhanced potential wetland inventory to what is expected using the extrapolated power law function (used to estimate the historical wetland inventory) (modified after Serran and Creed 2016).

topographically detectable on the landscape, namely those that have been filled or paved and are therefore permanently lost.

3.3 Identifying Restorable Wetlands

Ditch-drained wetlands were identified by developing an automated method to identify drainage ditch features on the landscape using a DEM. A flow chart of the method developed to identify restorable wetlands is shown in Figure 3.4 and consists of curvature analysis, reclassification, noise removal, and location filters to produce a final inventory of restorable wetlands.

3.3.1 Curvature

The identification of drainage ditches hinges on their topographic properties, namely their concavity. The LiDAR bare earth DEM with a horizontal resolution of 1 m and a vertical accuracy of 15 cm was used as the main input for the method. The DEM was not hydrologically-conditioned (i.e., pits were not filled) as the drainage ditches could be filled and therefore removed in the conditioning process. The curvature of the surface was calculated using the curvature tool is ESRI’s ArcGIS, which calculates the second derivative of the surface. Positive values indicated convex features, negative values indicated concave features, and flat surfaces had a curvature value of zero. An example of the curvature calculation is provided in Appendix A.

After applying the curvature tool to emphasize concave features, a low pass filter was then applied to smooth the data and remove noise, further emphasizing drainage features.

3.3.2 Reclassification

The smoothed curvature data were then reclassified to isolate the drainage ditches. Given that concave features are negative, an upper threshold was set to separate potential drainage ditches from other features. Jenks classification, an iterative variance minimization classification (Jenks 1967), was used to identify an appropriate break in the data associated with drainage ditch features. The chosen threshold, -6.61, was a stable break point, consistently appearing when data were binned into 4 or more classes. The data were reclassified to a binary map, with smoothed curvature values less than or equal

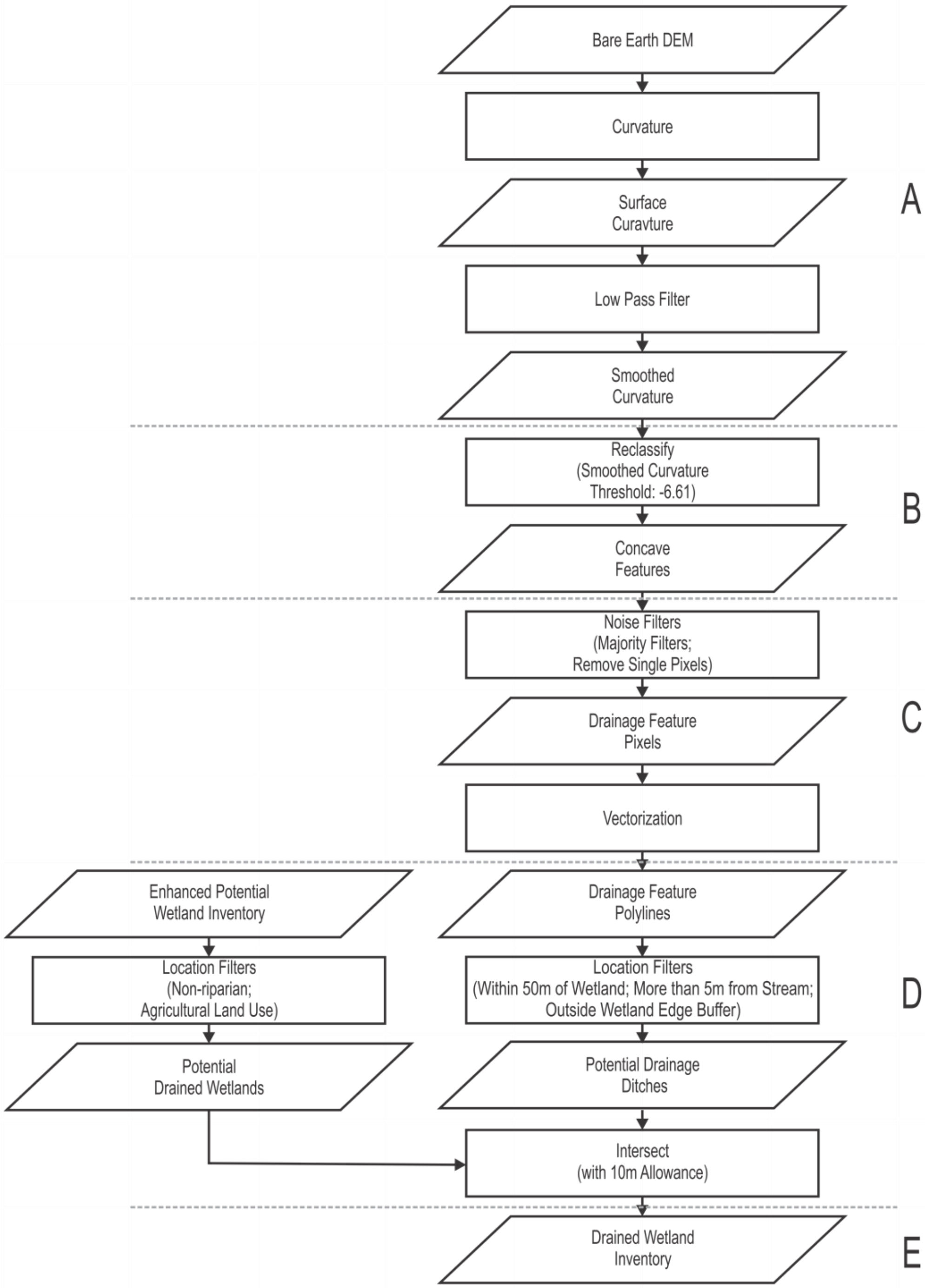

Figure 3.4 Flow chart of steps to delineate drained wetlands: (A) the surface curvature quantifies the convexity or concavity of a surface; (B) curvature reclassification narrows down the area of interest to concave features; (C) noise filters remove features that are not of interest including single pixels and features resulting from roads and railways; (D) location filters reduce potential drainage ditch candidates to those that are near wetland boundaries; and (E) the intersection of drainage ditch candidates with non-riparian wetland on agricultural land identifies ditch drained wetlands.

to -6.61 (representing potential drainage ditches) reclassified as “1”, and all other values (representing other surfaces) reclassified to “0”. The statistically-based threshold was also supported by digital terrain analysis data that were collected to understand the curvature values of concave features on the landscape. The smoothed curvature values for 100 drainage ditches and 100 furrows were sampled. Compared to other natural breaks identified using the Jenks method, a threshold of -6.61 minimized the inclusion of furrows with potential drainage ditches (data not shown).

3.3.3 Noise FIlters

The potential drainage ditches were filtered to remove features that were not of interest in the context of restorable wetlands. Drainage ditches are often located along roads and railroads, and while these features may in fact be draining wetlands, permanent infrastructure is unlikely to be altered to restore wetlands. Therefore, these drainage features were removed from consideration by creating a 15 m buffer on each side of roads and railroads and re-assigning the ‘potential drainage ditch’ pixels within these buffers to the ‘other’ class. Noise in the form of single pixels was then removed by applying two majority filters, which assign pixels a value based on the majority of the immediate neighbouring pixels, followed by the region group tool, which identifies isolated pixels that are not connected to pixels of the same class. The expand tool was applied to join adjacent drainage ditch features. Some drainage ditch features were broken up into pieces due to spatial variation in the smoothed curvature values along the feature, and the expand tool was used to grow the potential drainage ditch features by 1 pixel (1 m) in each direction, creating more continuous features. The potential drainage ditch features were then vectorized, that is converted from pixels to lines, which allowed for analysis of their spatial relation to other lines, namely wetland boundaries.

3.3.4 Location Filters

The potential drainage ditch lines were filtered based on their location. Drainage ditches are likely to cross or be adjacent to wetland boundaries, therefore, lines more than 50 m from an enhanced potential wetland were removed from further consideration. In addition, to remove natural drainage features, lines completely within 5 m stream buffers were also removed. A distance of 5 m was chosen because streams, which are often delineated using flow algorithms, do not always follow the current course of a stream, therefore an additional 5 m buffer acts to capture lines associated with streams. The intersection of drainage ditch lines with wetland boundaries posed a challenge. The change in slope that can occur along the boundaries of a wetland depression resulted in concave features, which needed to be removed. Using the enhanced potential wetland inventory, a 7.5 m buffer and a -7.5 m buffer around wetland boundaries were merged, creating a 15 m buffer which was centered on the enhanced potential wetlands. Drainage ditch lines that fell completely within these boundaries were eliminated. A 15 m buffer was chosen heuristically to balance the removal of concave features resulting from wetland edges while minimizing the elimination of drainage ditches. The remaining curvature features were buffered by 5 m to create polygons, which were once again converted to lines. Similar to the expand tool used previously, this aided in connecting discontinuous drainage ditch line features.

The enhanched potential wetlands were also filtered by their location to identify those most likely to contain drainage ditches. Only wetlands within agricultural land use areas were considered, as the practice of ditch-draining wetlands is largely associated with agriculture. Within the agricultural areas, riparian wetlands, those adjacent to streams, were removed as these wetlands are already connected to the drainage network and therefore not drained. The Agriculture and Agri-Food Canada (2013) annual crop inventory was used to identify agricultural land use.

The filtered drainage ditch lines and wetland boundaries were intersected to identify drained wetlands. Drained wetlands were identined as any non-riparian, agricultural wetland which was within 10 m of a potential drainage ditch. The 10 m distance allowance served as a precautionary safety-net to capture drainage ditches that lay just outside the periphery of wetland boundaries. Ten metres was chosen because initial accuracy assessment following the use of a simple intersect with no distance allowance resulted in some drained wetlands not being captured because the drainage ditch intersection was just short of the wetland boundary. The distance gap was on average 9 m, ranging from less than 1 m to 30 m, and the 10 m distance allowance reflected a balance between including ditch-drained wetlands and excluding other wetlands. The addition of a distance allowance around delineated wetland depressions also allowed for fluctuations in the wetland boundaries due to wet-dry cycles (Winter 1989).

3.3.5 Accuracy Assessment

One hundred random wetlands classified as ditch-drained drained (i.e., non-riparin wetlands within 10 m of a drainage ditch) and a 100 random wetlands classified as not ditch-drained (i.e., non-riparian wetlands more than 10 m from a drainage ditch) were sampled for an accurracy assessment of the method for detecting ditch-drained wetlands. The random samples were generated using the Subset Features tool in ArcGIS which uses a random number generator to extract a random subset of polygon features. To avoid propogation of errors, only wetland objects in the potential wetland inventory that showed evidence of being a wetland (see Section 3.1.5) were included in the sample sets. In addition, wetland objects that were dugouts were manually excluded from the sample as these human made features are not the focus of wetland restoration efforts. Using the DEM and historical and contemporary imagery, the samples were assessed for the presence of a drainage ditch feature. Similar to the wetland inventory accuracy assessment (section 3.1.5), the accuracy assessment was determined by generating a confusion matrix and calculating overall accuracy, producer’s and user’s accuracy, and Cohen’s kappa.

3.4 Understanding Wetland Loss

The historical wetland inventory, the enhanced potential wetland inventory (temporarily lost and existing wetlands), the restorable wetland inventory (ditch-drained temporarily lost wetlands), and the existing wetland inventory were compared to gain an understanding of the fate of wetlands within the watershed (see Table 3.3, Figure 3.5). For the existing wetland inventory (i.e., the CWI), wetlands that were identified as ditch-drained but that existed in the CWI were removed. The permanent historical loss was calculated as the difference between the historical inventory and the enhanced potential inventory. The temporary loss was calculated as difference between the enhanced potential wetland inventory and the existing wetland inventory. The restorable loss was

Table 3.3 Glossary of wetland inventory terms and wetland loss terms.

Term	Meaning
Enhanced potential wetlands	Potential wetland inventory merged with the CWI
Existing wetlands	CWI wetlands minus any wetlands identified as drained
Historical wetlands	Enhanced potential wetlands with the estimate of permanent wetland loss derived from the deviation from the power law
Permanent loss	Historical wetlands minus enhanced potential wetlands; the estimate of loss derived from the power law trend
Potential wetlands	Wetlands resulting from the object-based segmentation mapping method
Restorable wetlands/ Restorable loss	Number and area of wetlands identified as ditch-drained; a subset of temporary loss
Temporary loss	Enhanced potential wetlands minus existing wetlands; those wetlands that are not captured in existing inventory but whose depression is still present
Total loss	Permanent loss plus temporary loss; equivalent to historical wetlands minus existing wetlands

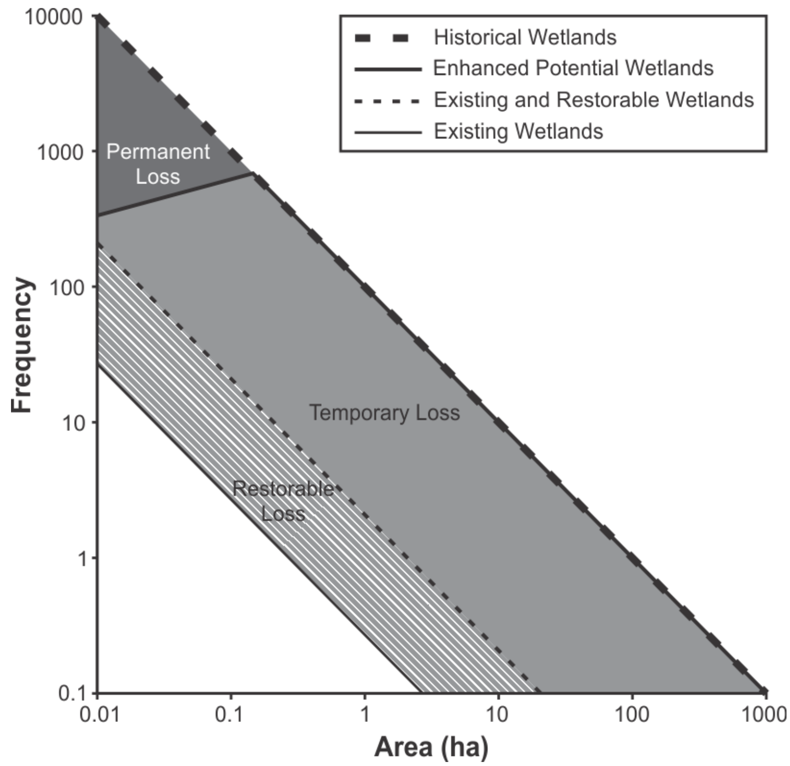

Figure 3.5 Concept of how the historical wetland inventory, enhanced potential wetland inventory, restorable wetland inventory, and existing wetland inventory can be compared to understand the number and area of permanent loss, temporary loss, and of the temporary loss, the portion that is restorable.

calculated as the portion of the temporarily lost wetlands that were ditch-drained. The pattern of wetland loss for different wetland sizes was also examined by comparing wetland size frequencies in the historical and existing wetland inventories. Additionally, for historical wetlands that were not completely lost, the sum of the existing wetland area and the number of existing wetlands within historical wetland boundaries was also examined.

Chapter 4

4 Results

4.1 Potential Wetland Inventory

Visual examples of each step of the potential wetland mapping method that was developed is presented in Figure 4.1. The potential wetland mapping method had a MMU of 0.0081 ha and identified 24,570 wetlands with a total area of 12,166 ha, which was 13.7% of the watershed area. The potential wetlands included 286 wetlands in developed areas with a total area of 358 ha, and 24,284 wetlands in non-developed areas with a total area of 11,809 ha. Of the wetlands in non-developed areas, 4,887 inundated areas with a total area of 1,996 ha were mapped within their wetland boundaries.

An accuracy assessment for the classification of the potential wetlands is presented in Table 4.1. The overall accuracy was 85.0% and the Kappa coefficient was 0.70. Of the areas classified as a wetland, 73.0% of the objects were confirmed to be wetlands from recent imagery. In the remaining 27.0% of cases, there was not enough evidence from the imagery to determine whether or not the object was a wetland (Figure 4.2A). Of the areas classified as other (non-wetland), 97.0% of the objects showed no evidence of wetland presence. In the remaining 3.0%, there was evidence of wetlands (Figure 4.2B). These omitted wetlands were not captured because their p_{dep} values fell below the threshold that was used to define a potential wetland, or because they fell within developed areas and there was no inundated area during the capture date of the NIR imagery.

The potential wetland inventory was able to capture 6,119 (or 89.2%) present wetlands mapped using the CWI. Of the 739 present wetlands which not captured, 285 present wetlands fell within developed areas. The 285 present wetlands within developed areas were not captured within the potential wetland inventory because they either did not have open water when the NIR image was captured, or it is also possible that they have been filled since 2006. An additional 69 present wetlands which were not captured did at one point have a wetland object within the potential wetland mapping method, however,

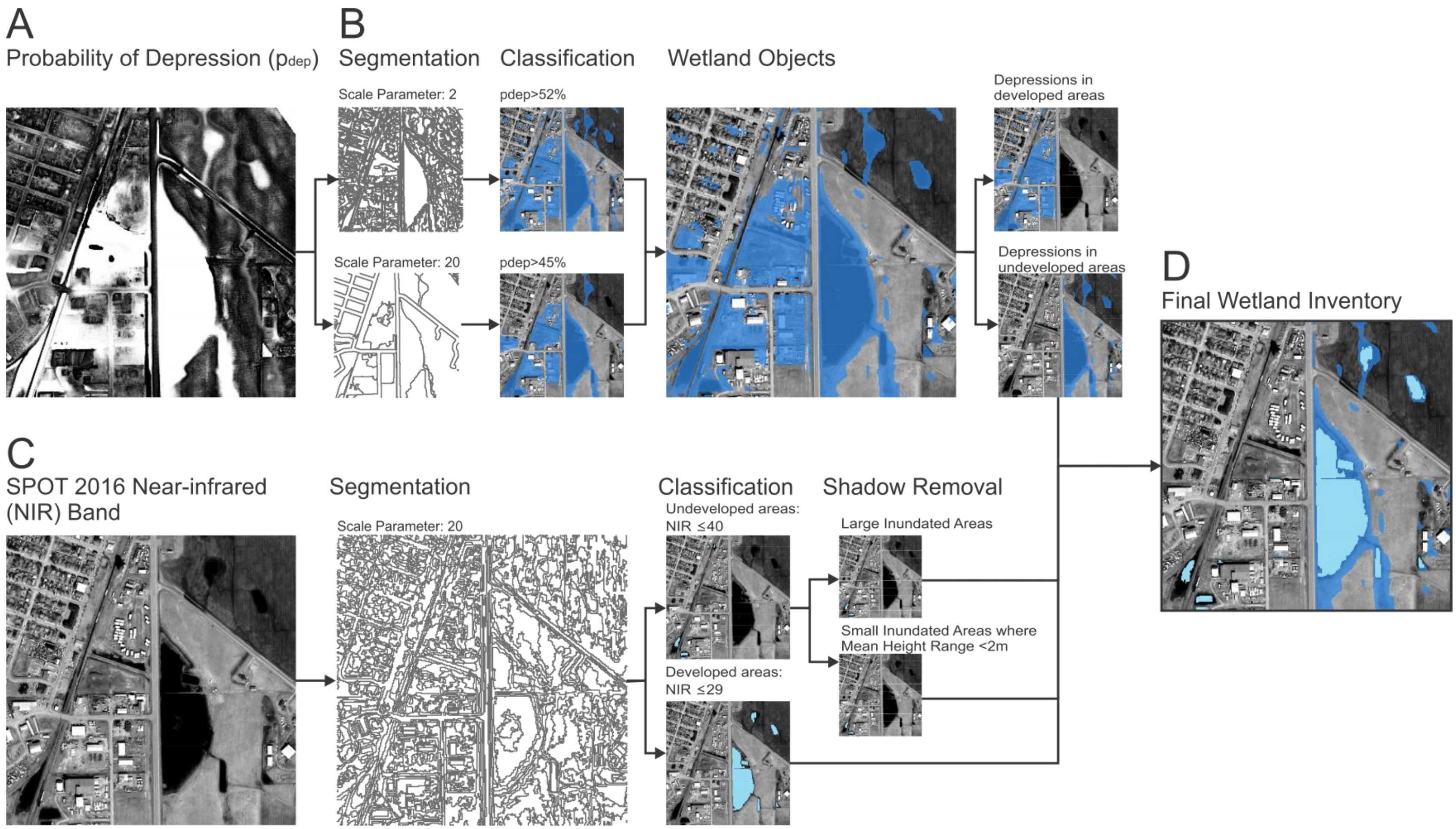

Figure 4.1 Images showing steps involved in delineating wetland depressions including: (A) stochastic analysis to identify the probability of depression; (B) object-based segmentation and classification of the probability of depression data; and (C) object-based image segmentation and classification of near-infrared imagery to produce (D) a final inventory of wetland depressions (dark blue) and inundated areas (light blue). Image centroid location: 51.4235°, -114.0252°.

Table 4.1 Accuracy assessment for wetland classification.

Class	Reference Totals	Classified totals	Number correct	Producer's accuracy (%)	User's accuracy (%)
Wetland	76	100	73	96.1	73.0
Other (Non-wetland)	124	100	97	78.2	97.0
Totals	200	200	170	-	-

Figure 4.2 Images showing examples of types of error for the wetland mapping method: (A) For this wetland object, there was no evidence within the available imagery that this object was in fact a wetland (commission error) (Image centroid location: 51.2160°, -114.0972°); (B) A wetland is visible in the available imagery, however, the automated technique for mapping wetland depressions did not indicate a wetland (omission error) (Image centroid location: 51.3919°, -114.1567°).

the wetland object did not meet the minimum size threshold. The remaining 385 present wetlands which were not captured within the potential wetland inventory were shallow, and therefore the associated p_{dep} values did not meet the p_{dep} thresholds.

4.2 Historical Wetland Inventory and Permanently Lost Wetlands

The enhanced potential wetland inventory (Figure 4.3A) included 20,027 wetlands with a total area of 12,498 ha within the entire watershed, and 19,753 wetlands with a total area of 12,361 ha in the undeveloped areas of the watershed. When applied to the undeveloped areas of the watershed, the relationship between wetland area and wetland frequency revealed a deviation from the power law reflecting the permanent loss of 2,451 wetlands with an area of 69 ha that were small (< 0.052 ha) (Figure 4.4). Therefore, the historical wetland inventory within undeveloped areas was estimated to be 22,204 wetlands with a total area of 12,431 ha.

4.3 Temporarily Lost Wetlands

The enhanced potential wetland inventory compared to the existing wetland inventory (Figure 4.3B) identified a temporary loss of 13,571 wetlands with an area of 9,732 ha. The proportion of temporarily lost wetlands with drainage ditches that make them easily restorable was 11.7% by number and 12.5% by area.

Visual examples for each step of the ditch-drained wetland mapping method is presented in Figure 4.5. Of the 11,279 non-riparian agricultural enhanced potential wetlands with a total area of 3,060 ha, 1,588 wetlands with a total area of 1,220 ha were classified as ditch-drained (Figure 4.3C). Therefore, 14.1% of the number and 39.9% of the area of the non-riparian agricultural enhanced potential wetlands were ditch-drained.

An accuracy assessment of the classification of restorable wetlands is presented in Table 4.2. Overall accuracy was 76.0% and the Kappa coefficient was 0.52. Of the wetlands classified as ditch-drained, 65.0% of the wetlands had a drainage feature present. In the remaining 35.0% of cases, misclassification was due to other concave features resulting from berms, the bottom of hills, furrows, or wetland edges (Figure 4.6A). Of the wetlands classified as not ditch-drained, 87.0% of the objects did not have a drainage feature present. In the remaining 13.0% of cases, misclassification was due to

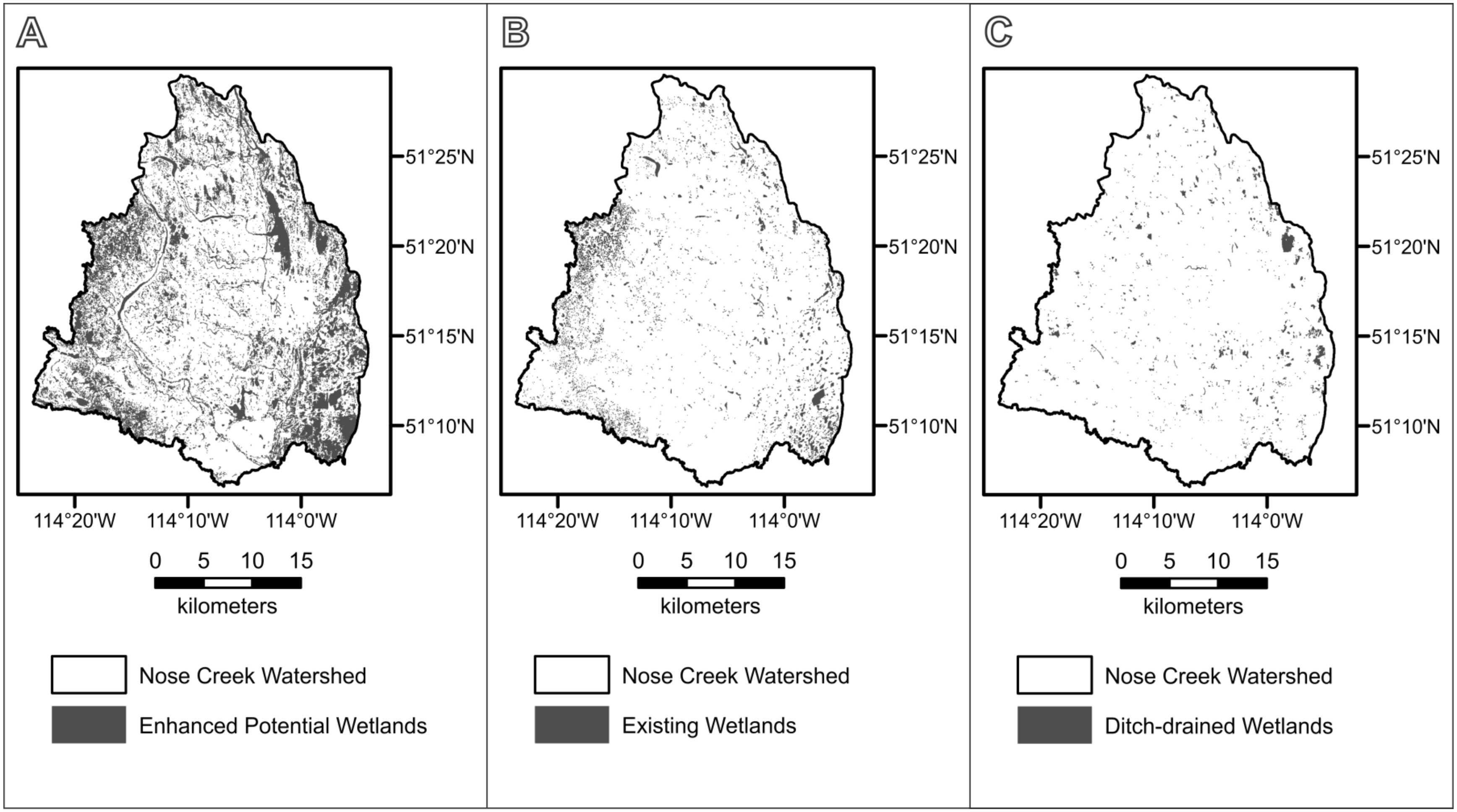

Figure 4.3 Maps of (A) enhanced potential wetlands, (B) existing wetlands, and (C) ditch-drained wetlands within the Nose Creek watershed.

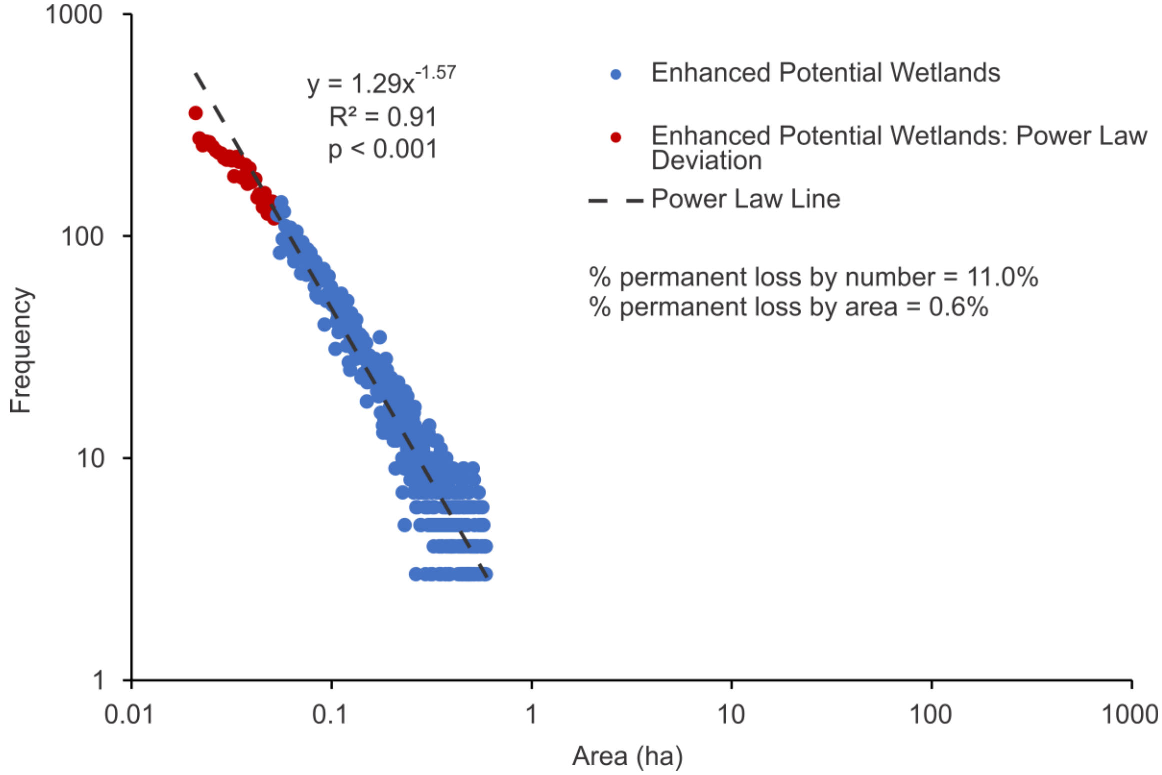

Figure 4.4 Plot of wetland area vs. frequency in logarithmic-logarithmic scales for the enhanced potential wetland inventory. Linear piecewise regression identified a deviation from the power law function which is used to estimate permanent loss and the historical wetland inventory. The deviation from the extrapolated power law line in the enhanced potential wetland inventory is shown in red and included wetlands less than 0.052 ha.

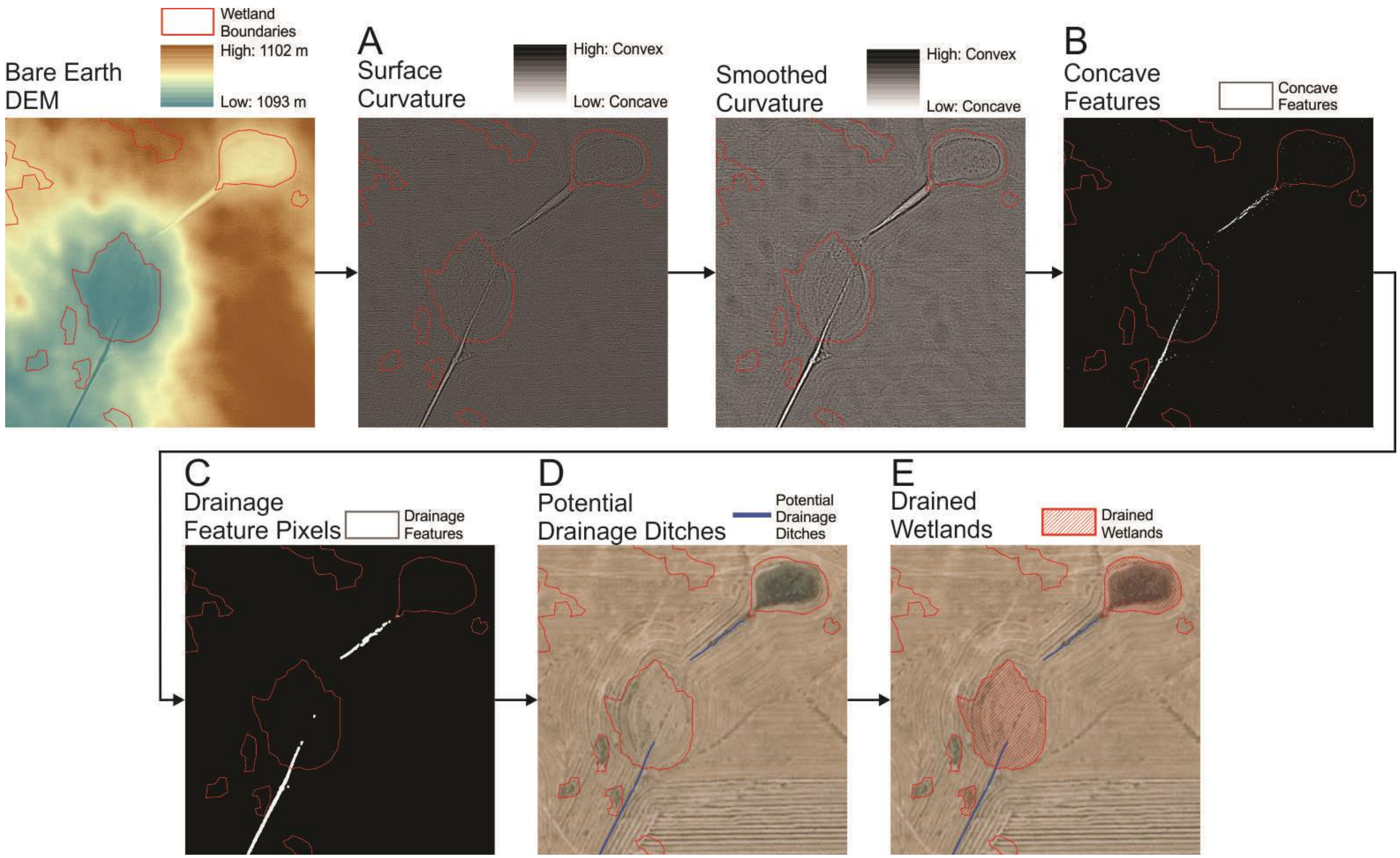

Figure 4.5 Images for the ditch-drained wetland mapping method. A bare earth DEM is used to calculate (A) surface curvature which quantifies the convexity or concavity of a surface. After smoothing the surface curvature, (B) reclassification narrows down the area of interest to concave features. (C) Noise filters remove features that are not of interest (single pixels, convex features from roads and railways). (D) Location filters narrow down the potential drainage ditches based on their spatial relationship with wetland boundaries. The intersection of drainage ditch candidates with non-riparian wetlands on agricultural land identifies (E) drained wetlands. Image centroid location: 51.2218°, -113.9186°.

Table 4.2 Accuracy assessment for ditch-drained or undrained wetlands.

Class	Reference Totals	Classified totals	Number correct	Producer's accuracy (%)	User's accuracy (%)
Ditch-drained wetland	78	100	65	83.3	65.0
Undrained wetland	122	100	87	71.3	87.0
Totals	200	200	152	-	-

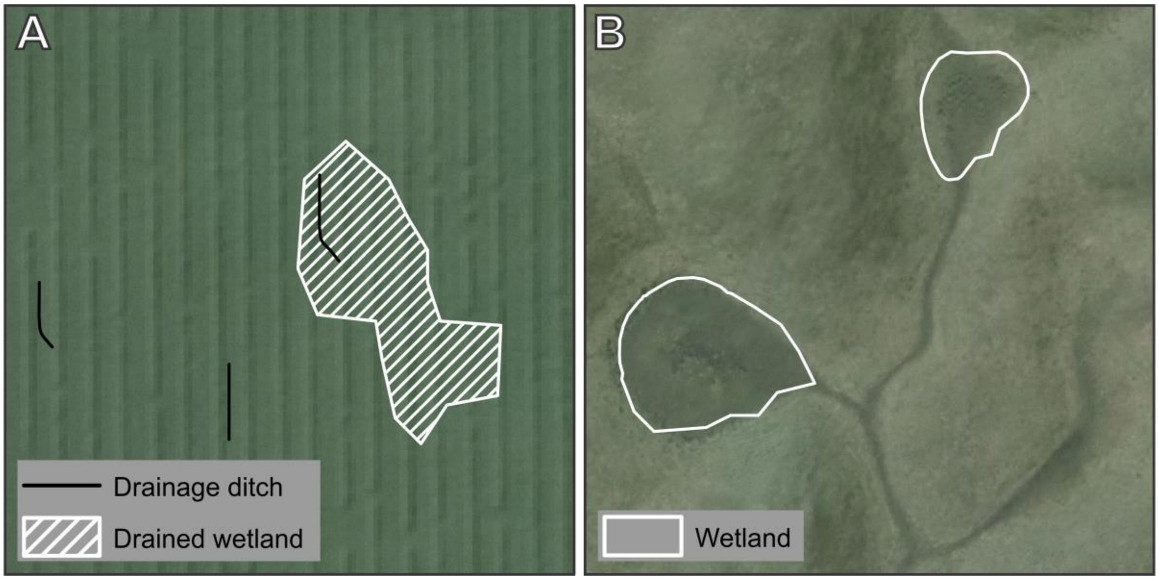

Figure 4.6 Images showing examples of the types of error for the ditch-drained wetland mapping method: (A) Concave features such as furrows were confused as drainage ditches, resulting in misclassification as drained (commission error) (Image centroid location: 51.3743°, -114.1756°); (B) Ditch-drained wetlands were misclassified as undrained when shallow ditches were not captured by the automated technique for mapping ditches (omission error) (Image centroid location: 51.3108°, -114.2802°).

ditch features being too shallow and therefore no curvature feature was detected, or a curvature feature was present but it was more than 10 m from the wetland (Figure 4.6B).

4.4 Area vs. Frequency Plots of Lost, Restorable, and Existing Wetlands

The area vs. frequency distributions for the historical, enhanced potential, restorable and existing wetland inventory in the undeveloped areas of the watershed is presented in Figure 4.7. By considering the wetland inventories together, absolute and proportions of wetland loss were calculated, presented in Table 4.3 and in Figure 4.8. Of the historical wetlands, only 27.8% of wetland numbers remain on the landscape, or only 21.2% of wetland area. There is an opportunity to increase existing wetland numbers by 25.7% (from 6,182 wetlands to 7,770 wetlands) and wetland area by 46.4% (from 2,630 ha to 3,850 ha) through restoration of ditch-drained wetlands.

The percent total wetland number loss vs. wetland size presented in Figure 4.9 shows a trend of substantial loss (> 75%) of both small (especially < 0.32 ha) and large (> 0.82 ha) wetlands. The least loss, about 50%, occurs for wetland sizes between 0.32 ha and 0.82 ha. In absolute numbers, small wetlands (<0.32 ha) have seen a total loss of 13,017 wetlands by number and 1,077 ha by area, and large wetlands (>0.82 ha) have seen a total loss of 2,224 wetlands by number and 10,241 ha by area. Of the historical wetlands that have not been completely lost, by looking at the sum of the existing wetlands area within historical wetland boundaries, historical wetlands are seen to be getting smaller (Figure 4.10). Similarly, by looking at the number of existing wetland objects within historical wetland boundaries (Figure 4.11), historical wetlands are seen to be breaking up into pieces, with the largest historical wetlands becoming the most fragmented.

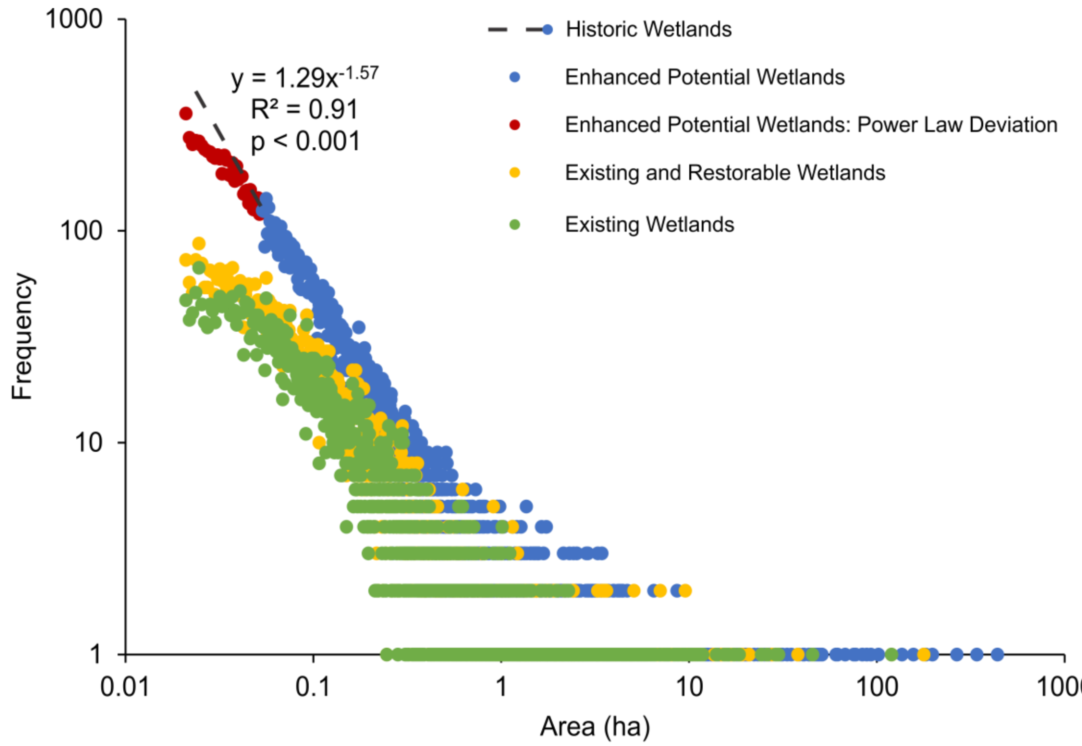

Figure 4.7 The area vs. frequency distributions in logarithmic-logarithmic scales for the historical wetland inventory, existing and restorable wetland inventory, and the existing wetland inventory. A comparison of the inventories identifies permanent loss, temporary loss, and the portion of temporary loss that is restorable (also see Table 4.3).

Table 4.3 Absolute number and percent of historical wetlands, permanent loss, temporary loss, restorable loss, and existing wetlands in the undeveloped areas of the Nose Creek watershed.

	Absolute number	**Percent of Historical Wetland Total**
Historical wetland number	22,204	100.0%
Historical wetland area (ha)	12,431	100.0%
Permanent loss by number	2,451	11.0%
Permanent loss by area (ha)	69	0.6%
Temporary loss by number	13,571	61.1%
Temporary loss by area (ha)	9,732	78.3%
Restorable loss by number	1,588	7.2%
Restorable loss by area (ha)	1,220	9.8%
Existing wetland number	6,182	27.8%
Existing wetland area (ha)	2,630	21.2%

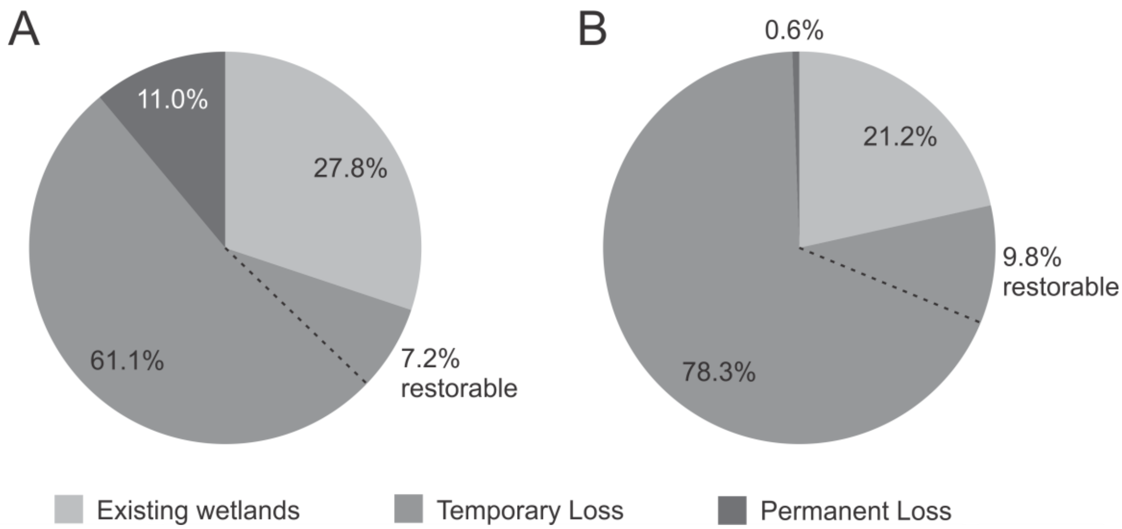

Figure 4.8 Pie charts showing the percent (A) by number, and (B) by area of historical wetlands in the undeveloped areas of the watershed which are permanently lost, temporarily lost, and existing. Under temporary loss, the percent of historical wetlands which are restorable is also shown.

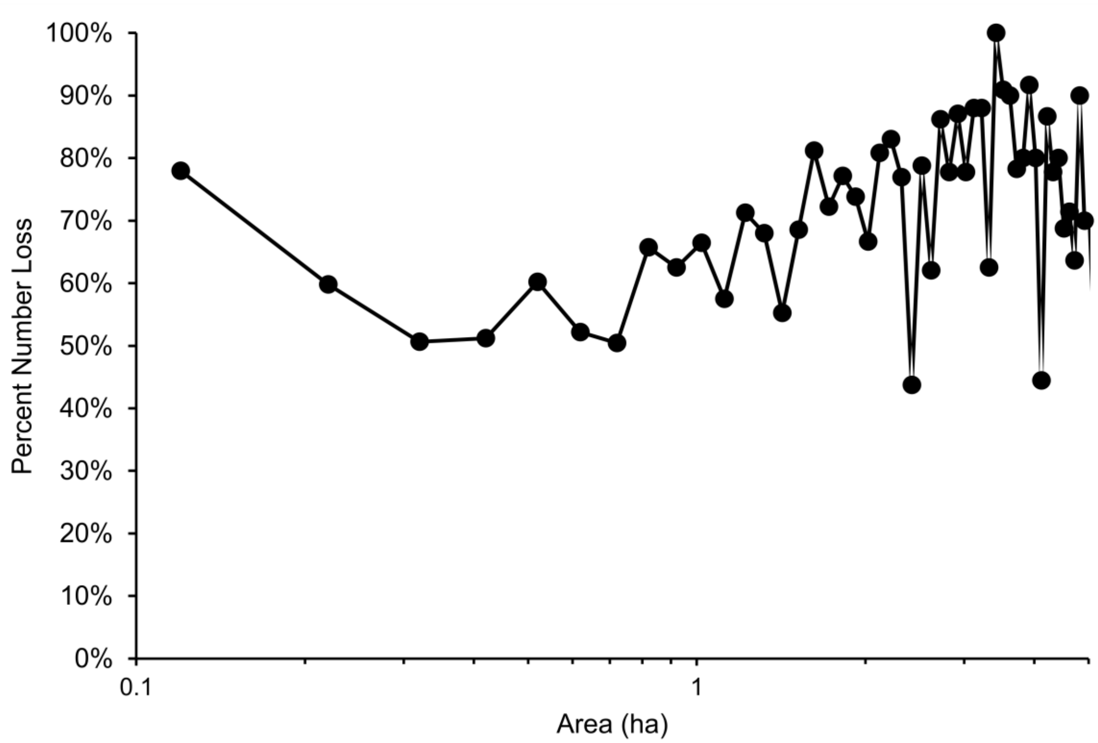

Figure 4.9 Percent total wetland loss by number for different wetland sizes (includes permanent loss and temporary loss).

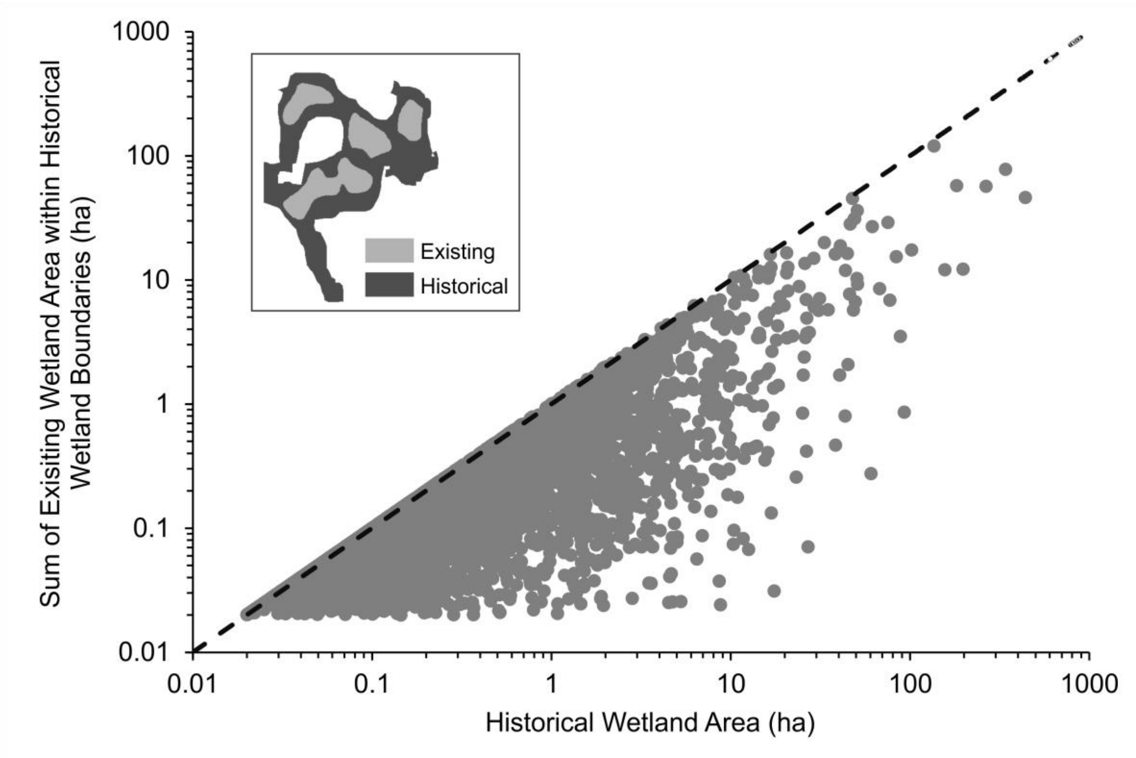

Figure 4.10 The sum of existing wetland area within historical wetland boundaries, as a function of historical wetland area, in logarithmic-logarithmic scales. The dashed black line shows a 1:1 relationship. The inset image shows an example of a wetland located at 51.1688°, -114.2553°.

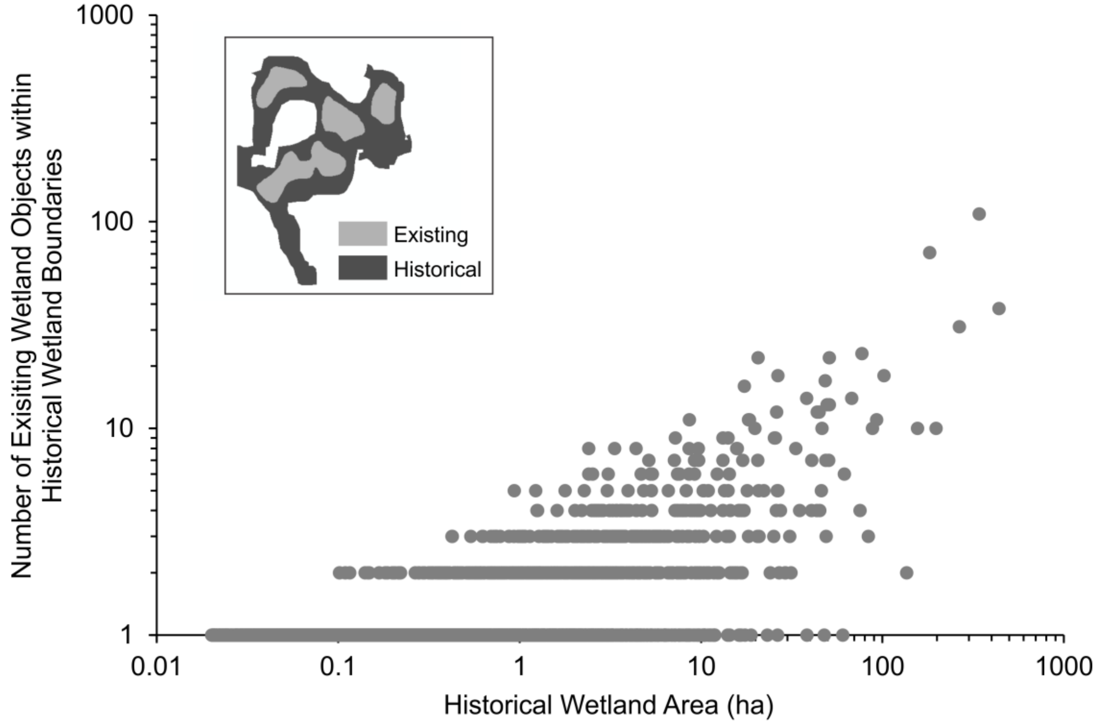

Figure 4.11 The number of exiting wetlands objects within historical wetland boundaries, as a function of historical wetland area, in logarithmic-logarithmic scales. The inset image shows an example of a wetland located at 51.1688°, -114.2553°.

Chapter 5

5 Discussion

Comprehensive wetland inventories that demonstrate the magnitude of existing, restorable and not easily restorable wetlands are required by wetland managers to help prioritize restoration efforts. Automated wetland mapping methods that provide inventories of wetlands that exist, that have been lost, and that are easy to restore are needed to provide tangible launch points for sustainable wetland management.

5.1 Potential Wetland Inventory

An automated method for potential wetland mapping was developed, building on Serran and Creed (2016). The major advancement from the Serran and Creed (2016) method was to improve wetland mapping in developed areas. The previous multi-segmentation method used topographic information only to identify potential wetlands. This method was improved to delineate inundated (open water) areas within topographic depressions using NIR imagery. Within developed areas, inundated area boundaries were used instead of topographic depressions to better reflect the wetlands on the landscape. The revised method is now more automated and does not require manual manipulation in developed areas, which further increases its application efficiency. An additional strength of mapping inundated areas is that they can be further used to establish restoration potential. Hydrology is one of the most important considerations in restoration efforts (Zedler 2000), and agricultural activity over many decades can change soils and water table levels (Zedler 2000, 2003). Therefore, wetland inventories which include inundated areas provide information on current hydrology and whether water is likely to return, aiding in restoration decision making.

One limitation of this method is that it does not consider hydric soil data in establishing wetland presence (McCauley and Jenkins 2005; Van Meter and Basu 2015), and therefore wetland areas can be potentially overestimated. The inclusion of hydric soil data can help remove depressions from the inventory that are not wetlands. Nonetheless, with an overall accuracy of 85.0%, the method is considered useful. The accuracy could have been higher had there been a longer time series of imagery to provide empirical evidence of inundated areas (which can fluctuate among seasons and across years).

5.2 Historical Wetland Inventory

The historical distribution of wetlands can be difficult to determine, given that they are often altered, and therefore more difficult to detect on the current landscape. A historical time series of imagery can be used to create wetland inventories through time, however, this requires a significant amount of data and analysis (Ozesmi and Bauer 2002), and this assumes that the imagery is available at the temporal and spatial resolution needed. Hydric soil data can be used to estimate the historical extents of wetlands (Galatowitsch and van der Valk 1994; Miller et al. 2009), however, soil data, when available, is often coarse in resolution. This poses a problem for prairie potholes that are often smaller than the available soil data resolution.

The automated method of identifying historical wetland extent was chosen due to several strengths. First, historical wetland estimates could be identified based on contemporary landscapes. Second, the power law line in the area vs. frequency plot was used to capture permanently lost wetlands – those wetlands that have been filled and whose depressions are no longer detectable. Third, the combination of enhanced potential wetland inventory, ditch-drained wetland inventory, and existing wetland inventory was used to capture temporary wetland loss – those wetlands that are not intact or ditch-drained, but whose wetland basins are still detectable. These strengths are particularly advantageous in areas where historical data are not available. The continually improving technologies to capture topography is resulting in the increased availability of finer resolution data both in space and time (Knight et al. 2015). The accessibility of fine resolution data means the automated tool has the potential to be applied broadly to provide historical estimates of wetland extent as well as permanent and temporary loss. In addition to identifying wetlands, LiDAR data also allows for wetland characterization and classification by providing information on vegetation (Rosso et al. 2006; Gilmore et al. 2008), and water flow and storage (Lindsay et al. 2004; Lane and D’Amico 2010; Huang et al. 2011; Knight et al. 2013). LiDAR data can also be combined with emerging airborne and satellite remote sensing technologies (Töyrä and Pietroniro 2005; Moffett and Gorelick 2013; Huang et al. 2014; Lang et al. 2015), further improving and enhancing wetland identification and characterization capabilities.

One limitation of the historical wetland inventory is that the identification of permanently lost wetlands is non-spatial. However, permanently lost wetlands are not likely candidates for restoration, and therefore there is no need to have maps of them for restoration activities.

5.3 Restorable Wetland Inventory

The automated method for mapping restorable wetlands in this thesis was based on a high resolution digital terrain model and targeted a specific mechanism of wetland loss, that is wetlands with drainage ditches that can be plugged. Targeting ditch-drained wetlands is a major strength as it identifies wetlands that can be easily restored. Surface curvature has been successful in identifying drainage ditches in other environments (Pirotti and Tarolli 2010; Passalacqua et al. 2012). The method developed in this study is simple and replicable with an acceptable overall accuracy of 76.0%. However, the user's accuracy for identifying drained wetlands (65.0%) can be a barrier for restoration practitioners, as it still leaves false positives to filter through. Misclassification was due to confusion with other concave features, including furrows. For practitioners looking to restore either ditch-drained or cultivated wetlands, the user classification would increase to 78.0%, as 13.0% of wetlands were misclassified as drained due to furrows. While the tool does not distinguish between natural and human made ditch features, natural drainage pathways can be human modified (Watmough and Schmoll 2007), and are therefore appropriate to include in the inventory for further investigation.

The automated method for mapping ditch-drained wetlands can be improved by implementing additional automated steps to remove unwanted features. In particular, dugouts were manually removed during accuracy assessment as they often have adjacent berms, the bottom of which are mistaken for ditch features. Dugouts consistently have standing water, and generally have a rectangular shape (Alberta Agriculture and Food 2007). Dugouts can potentially be automatically removed from consideration by considering the perimeter to area ratio of inundated area features. Additionally, the tool can be further tailored to the needs of restoration practitioners by including aspects relating to restoration feasibility, such as accessibility and size constraints.

5.4 Proportion of Lost, Restorable, and Existing Wetlands

This study has sought to highlight the potential for wetland recovery by categorizing and quantifying permanent wetland loss, temporary wetland loss, and restorable wetland loss. Previous studies have used similar methods to identify temporary loss (Van Meter and Basu 2015), or permanent loss (Serran and Creed 2016), however, this study builds on these previous methods to present a comprehensive inventory of wetland loss directed toward wetland management decision-making.

There are remarkably few detailed estimates of wetland loss for the region against which to compare these results (Dahl and Watmough 2007). Wetland loss across the PPR have been reported to be between 40-70% since settlement (Schick 1972; Lynch-Stewart 1983; Rakowski and Chabot 1984; Environment Canada 1986; Glooschenko et al. 1993; Strong et al. 1993; Rubec 1994; Alberta Environment 1996; Dahl and Watmough 2007); however, many reports of loss lack vital details of how loss is defined, how estimates were derived, what the minimum size of wetland that is considered, and whether loss is reported by number or area. Several wetland loss estimates commonly cited for the region are also derived from unpublished reports (e.g. Schick 1972; Goodman and Pryor 1972; Rakowski and Chabot 1984; Strong et al. 1993), making this knowledge on wetland loss inaccessible and possibly not peer reviewed. Working in the Alberta aspen parklands, Schick (1972, as cited in Lynch-Stewart 1983) made use of township survey plans, government drainage districts, and aerial photographs to assess wetland change between 1900 and 1970 and found a 61% loss of wetland area. Estimates of 40-70% loss are below this study’s estimate of 72.2% total wetland loss by number and 78.8% total loss by area, which may be due to continued wetland loss since previous estimates were made.

Some estimates of wetland loss are based on wetland inventories derived from high resolution LiDAR data. Using a similar approach to the one developed in this thesis but with a larger MMU (i.e., 0.04 ha compared to 0.02 ha, respectively), Van Meter and Basu (2015) generated a historical wetland inventory using a 1 m DEM and 1:15,840 scale hydric soil data together with existing wetland estimates based on the 1:24,000 scale U.S. NWI. Van Meter and Basu (2015) estimated a 90% historical loss of wetland area in the Iowa portion of the PPR, which is more comparable to the loss estimates found for the Nose Creek watershed.

Other estimates of wetland loss are based on changes in wetlands between two points in time. For example, working in the black soil zone of Alberta, Saskatchewan and Manitoba, Goodman and Pryor (1972, as cited in Lynch-Stewart 1983) used aerial photographs, waterfowl capability maps, agricultural capability maps, soil surveys, and field surveys to sample 600 random quarter sections and found a 13% net loss of wetland area between 1940 and 1970. More recently, Watmough and Schmoll 2007 surveyed transects across the Canadian Prairies and found that over a 17 year period (between 1985 and 2001), wetland loss was estimated at 5%. Similarly, Clare and Creed (2015) used wetland inventories generated from aerial photographs over a 11 year period (between 1999 and 2009) and found 242 wetlands totalling 71 ha were lost in the Beaverhill watershed of central Alberta. Loss estimates between two points in time do not reflect historical loss, and therefore it is not possible to state the relatively importance of the wetland loss estimates from these studies – they only confirm that wetland loss is continuing.

Estimates of restorable wetlands are also difficult to come by. In their assessment of 600 random quarter sections in the Prairies, Goodman and Pryor (1972, as cited in Lynch-Stewart 1983) found that 19% of wetlands by area had been affected by drainage or partial filling. Schick (1972, as cited in Lynch-Stewart 1983) found 34% of wetland area had been lost to drainage between 1900 and 1970. In Nose Creek, 9.8% of historical wetland area was found to be drained, lower than these previous estimates. Working in the Minnesota and Iowa portion of the PPR using aerial photograph interpretation, the Restorable Wetland Working Group assessed 1,036,000 ha and found 1,500 drained wetlands (Ducks Unlimited n.d.). In comparison, in the Nose Creek Watershed, 1,588 drained wetlands were identified in the 63,217 ha agricultural area of the watershed. The spatial and temporal variability of the few estimates of drained wetlands available are very difficult to directly compare to the estimates found for Nose Creek. The sparsity and variability of estimates emphasizes the need for such automated tools as presented in this study that standardize the terminology and method of identifying lost and restorable wetlands.

5.5 Application of Loss Estimates to Wetland Management

The wetland loss estimates found in this study confirm the bleak reality of the fate of wetlands within the PPR. With the total loss of 72.2% of wetlands by number, and 78.8% of wetlands by area, also comes the loss of wetland ecosystem functions and associated services (Zedler and Kercher 2005). Current wetland loss has included the preferential loss of both small (<0.32 ha) and large (>0.82 ha) wetlands, leading to a homogenization of wetland sizes, with historical wetlands disintegrating into smaller fragments. This is particularly worrisome, as wetland size is a determinant of wetland ecosystem function (Creed and Aldred 2015). For example, small and isolated wetlands such as those found in the Prairies support species richness and biodiversity (Semlitsch and Bodie 1998; Leibowitz 2003; Scheffer and van Geest 2006). Furthermore, waterfowl populations are particularly vulnerable to wetland loss and size homogenization, as they depend on a variety of wetland types and sizes throughout their life cycles (Kantrud et al. 1989). Similarly, water quality is also affected, as biogeochemical functions such as sedimentation, denitrification, and phosphorus storage vary with wetland size and connectivity (Whigham and Jordan 2003; Marton et al. 2015; USEPA 2015). Functions such as water storage capabilities are also dependant on size, with the loss of wetlands leading to concerns of increased flooding (Miller and Nudds 1996; Gleason et al. 2007).

As wetland restoration and recovery efforts increase, wetland managers should bear in mind the trends found in this study. If the aim is to restore historical watershed scale patterns in wetland distribution, the loss of both small and large wetlands should be considered simultaneously in restoration efforts. Restoration efforts themselves can contribute to landscape homogenization when specific wetland types or sizes are favoured (Bedford 1999). Therefore knowledge of both restorable wetlands and historical wetland distributions are important for well-informed wetland management.

Restoration of the drained wetlands is likely to lead to a substantial increase in wetland ecosystem services in the Nose Creek watershed. Of the 11,279 non-riparian, agricultural wetlands in the watershed, approximately 14.1% were identified as drained, equivalent to 39.9% of the non-riparian, agricultural wetland area. This represents an important opportunity for re-establishing valuable wetland ecosystem functions. While restorable wetlands represent almost 9.8% of the historical inventory area, the proportion is still relatively small. Therefore, wetland management efforts should also consider restoration of wetlands lost through other means such as cultivation. Turner et al. (1987, as cited in Bethke and Nudds 1995) found that an approximately 40% of wetland basins between 1981 and 1985 in the Canadian Prairies were affected by cultivation, suggesting a high potential for recovery among cultivated wetlands.

Chapter 6

6 Conclusion

6.1 Research Findings

The objectives of this research were to: (1) advance an established automated wetland mapping method by incorporated NIR imagery to delineate wetlands within urban areas; (2) develop an automated ditch-drained wetland mapping method to identify restorable wetlands; and (3) use the wetland inventories to generate historical wetlands, existing wetlands, lost wetlands and among those that have been lost, potentially restorable wetlands within the Nose Creek watershed in southern Alberta.

The revised multi-resolution object-based segmentation method was successful at capturing a broad range in size of potential wetlands (i.e., wetland depressions that may be functioning as wetlands or not) both in natural and developed areas. Based on a 3 m LiDAR DEM, the potential wetland inventory with a MMU of 0.0081ha (81 m^2) contained 24,570 wetlands with a total area of 12,166 ha and an overall accuracy of 85.0%. The method does not require historical data or laborious aerial photograph or satellite image interpretation. A power law analysis of the area vs. frequency of these potential wetlands facilitated estimates of historical wetlands, permanently lost wetlands (i.e., either filled or paved over), and, together with the existing wetland inventory, temporarily lost wetlands (i.e., the depression remains, but it is no longer functioning as a wetland). Based on a MMU of 0.02 ha, the historical number and area of wetlands was estimated as 22,204 wetlands and 12,431 ha. The permanently lost wetlands was estimated to be 11.1% in number and 0.6% in area of this historical number, and the temporarily lost wetlands was estimated to be 61.1% in number and 78.3% in area of this historical number. Existing wetlands represented only 27.8% of the total historical number and 21.2% of the total historical area, reflecting a remarkable loss of wetlands as has been reported in other studies (Lynch-Stewart 1983; Van Meter and Basu 2015). Small (<0.32 ha) and large (>0.82 ha) wetlands have seen the largest percent loss. Wetland loss in the Nose Creek watershed has been substantial, and efforts to reverse this trend are crucial if the important ecosystem services provided by wetlands are to be maintained on this landscape.

The new automated ditch-drained wetland mapping method was also successful. Surface curvature was calculated from a 1 m LiDAR DEM to identify concave features on the landscape. Together with the enhanced potential wetland inventory, the relative location of a concave feature to a wetland was used to identify ditch-drained wetlands, which can be easily restored. When applied to the Nose Creek watershed, 1,588 restorable wetlands with an area of 1,220 ha were identified with an overall accuracy of 76.0%. The restorable wetlands were estimated to be 7.2% of the total historical number and 9.8% of the total historical area.

6.2 Scientific and Management Significance

Wetland inventories are the foundation of sustainable wetland management. The automated inventory methods presented here are simple, transparent and reproducible. The generated inventories facilitate a multi-faceted view of the fate of wetlands, estimating the extent of historical, permanently and temporarily lost, restorable, and existing wetlands. An understanding of this wetland change on a watershed scale, together with insights on changes in their distribution, can guide both protection and restoration efforts, and shape wetland and watershed management goals.

6.3 Further Research Directions

Many exciting opportunities exist to further the state of wetland research. For example, opportunities exist to identify other types of restorable wetland loss. While some wetland loss was accounted for as permanent (i.e. filled wetlands) or restorable (i.e. ditch-drained), the temporary loss of 61.1% by number remains to be further categorized. Research opportunities exist to bring together methods that investigate the mechanisms behind other forms of wetland loss, such as cultivation and subsurface drainage. For example, Naz et al. (2009) have used high resolution aerial imagery to map individual tile lies using edge detection filters. Cultivated wetland basins can also be identified through land use classification (e.g. Fenstermacher et al. 2014). Given the minimal influence of subsurface drainage that have been reported for the Canadian Prairies (Government of

Manitoba 1985), land use classification to identify wetlands lost to cultivation should be a next priority. This would provide an even clearer picture of wetland loss and shed light on the further restoration potential within the watershed.

Furthermore, automated tools to predict the potential functions of restored wetlands would be beneficial in prioritizing wetland restoration projects, as well as tailoring projects to the specific functional needs of a community (Zedler 2003). Such tools would be especially useful in Alberta, as the recent Alberta Wetland Policy (Government of Alberta 2013) has moved from area-based toward function-based assessments of wetlands when considering the fate of these wetlands. The foundation for such tools has already been established. For example, Creed and Aldred (2015) have developed an automated tool to estimate functions of existing wetlands. An important scientific contribution would be to extend this tool to estimate functions of potentially restorable wetlands so that managers can project the changes in ecosystem services provided by these wetlands into the future (e.g., Accatino et al. 2016). Knowledge of the distribution of existing and potential wetland functions across a landscape will enable more targeted management actions to achieve policy objectives. A restoration prioritization tool based on potential functions also provides a way to engage landowners and the public about the benefits of restoration, since the inability to communicate the value of wetlands can be a barrier to preservation and restoration (Lynch-Stewart 1983; Kauffman-Axelrod and Steinberg 2010).

Following restoration, monitoring is still required to determine if and how quickly different wetland functions can recover (Bartzen et al. 2010). An adaptive monitoring approach is required to study restoration results and apply the findings (Zedler 2003). Cost effective monitoring approaches have also not been established, however, emerging remote sensing technologies, including new sensors and unmanned aerial vehicles, are promising, reducing the need for expensive field work, and increasing accessibility (Jensen et al. 2011; Knoth et al. 2013; Gallant 2015).

Appendix A: Calculation of Curvature

Curvature was calculated using the curvature tool is ESRI's ArcGIS. Given a hypothetical raster grid with a resolution, L:

P1	P2	P3
P4	P5	P6
P7	P8	P9

The curvature for pixel P5 is calculated as:

$$\text{Curvature} = [(4P5-P2-P4-P6-P8)/L^2]*100$$

It should be noted that the curvature tool provides the equivalent results to the Laplacian filter, which is commonly used for edge enhancement and has been recommended in the identification of drainage ditches in low relief landscapes (Passalacqua et al. 2012). The Laplacian filter also approximates the second derivative of a surface using a kernel (Liu and Mason 2009):

0	-1	0
-1	4	-1
0	-1	0

When applying the Laplacian filter to pixel P5 of the hypothetical raster above, the resulting value is calculated as:

$$\text{Laplacian} = (4P5-P2-P4-P6-P8)$$

Given the resolution of this DEM (1 m), the results from ESRI's Curvature tool are essentially equal to those of a Laplacian filter, differing by a factor of 100.

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Curriculum Vitae

Name:	Ann Waz
Post-secondary Education and Degrees:	University of Western Ontario London, Ontario, Canada 2014-2016 M.Sc. University of Guelph Guelph, Ontario, Canada 2008-2013 B.Sc.Env.
Honors and Awards:	Natural Sciences and Engineering Research Council Scholarship (Masters) 2015-2016 Western Graduate Research Scholarship 2014-2015 School of Environmental Sciences Academic Prize 2013 Dean's Honours List 2008-2012
Related Work Experience:	**Teaching Assistant** – University of Western Ontario 2014-2016

Presentations:

Waz, A., Creed, I. F. (2016). Automating the identification of altered wetlands. Poster. International Association for Great Lakes Research (IAGLR) Conference, June 6-10, 2016, Guelph, Ontario.

Waz, A., Creed, I. F. (2016). Automating the Identification of Drained wetlands. Poster. Sustainagle Agricultural Systems in the Great Lakes Basin Symposium, April 14, 2016, London, Ontario.

Waz, A., Creed, I. F. (2016). Automating the Identification of Drained Wetlands in the Prairie Pothole Region. Poster. World Wetlands Day Symposium, March 22, 2016, Waterloo, Ontario.

Waz, A., Creed, I. F. (2015). Identifying Drained Wetlands and their Anticipated Functions in the Prairie Pothole Region. Poster. Canadian Network for Aquatic Ecosystem Services Annual meeting, April 29-30, Sault Ste. Marie, Ontario.

Waz, A., Creed, I. F. (2014). Cumulative effects of geographically isolated wetland loss in the prairie pothole region. Oral presentation. Canadian Network for Aquatic Ecosystem Services Theme II meeting, December 4-5, Toronto, Ontario.