University of Alberta

On the origin of virtual wetlands by means of computer aided selection
Or, the preservation of favoured places in the struggle for functional wetlands

by
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Dedication

I want to dedicate this to my Mom who gave me the confidence to pursue higher learning. I think she would have been proud.

Abstract

To aid in reclamation planning for the Genesee Coal Mine in Alberta, I qualified the pre-mined state of wetlands and measured land use and land cover (LULC) change between 1982 and 2007. A generalized linear model (GLM) was developed to explain the presence of wetlands on the pre-mined landscape. Environmental variables used to model the distribution of the wetlands included categorical LULC variables (agricultural land, vegetation, roads, structures, rivers, streams and tributaries), and elevation or elevation-derived terrain variables (slope, terrain ruggedness index, compound topographic index, sinks). Results from the model suggest that pre-mined wetland presence is best explained by agricultural land use, distance to tributaries, terrain ruggedness, distance to rivers, and the interaction between agriculture and roads. Landscape metrics were used to measure changes in landscape fragmentation and wetland structure. Differences in metric values suggest that the landscape has more surface water, less forested or vegetated land cover, and greater fragmentation.

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Chapter 1 Introduction

1.0 Research background and rationale

1.1 Equivalent capability

The government of Alberta requires resource extraction industries to minimize environmental impacts during and after operations by reclaiming the landscape to equivalent land capability under Alberta's Environmental Protection and Enhancement Act (EPEA). Reclaiming land to equivalent capability does not mean that land conditions have to be identical to pre-disturbed land conditions but rather that reclaimed lands have similar capabilities to pre-disturbed land or adjacent and similar land. Surface mining activities at the Genesee Coal Mine (herein Genesee) in Central Alberta have prompted mitigation for wetland loss and restoration of land cover that will sustain both ecological and agricultural land uses. Sustainability is important for maintaining environmental health and, more specifically, environmental goods (such as wetlands) and services (such as groundwater recharge) (Olewiler 2004). Spatial models are one approach to measuring land use and land cover (LULC) change and the effects on landscape sustainability. Land use and land cover are interrelated (Leitao et al. 2006) and are therefore integral to spatial analysis at Genesee because LULC are affected by agronomic activities (Reyes et al. 2008) and have a pronounced effect on water quantity/quality (Harper et al. 2008; Zampella et al. 2007). In the context of this study, restoration refers to the process of bringing back natural ecosystems (e.g. wetlands) to a prior state with the intent being “reinitiating ecosystem processes” whereas reclamation is used to refer to the process of returning land to a former production system (e.g. agriculture) (Clewell and Aronson 2007). Reclamation is the term more commonly used when referring to overall land use practices and post-mining activities at Genesee.

An analysis of LULC at the landscape level will help to determine the degree of success of restoration efforts and land capabilities at operational scales (multiple kilometers). Restoration of individual wetlands requires analysis of the surrounding landscape because wetland position, shape, elevation and adjacency to other habitat types all have implications for wetlands (Mitsch and Gosselink 2000). Using a landscape scale approach, the intent of this research is to assist reclamation endeavors at the Genesee mine site by developing alternative landscape designs that consider present day and adjacent land uses while identifying suitable sites for wetlands on reclaimed lands. The first goal of this research is to assess and qualify the placement of wetlands on the pre-mined landscape. The second goal is to assess and qualify the composition and configuration of wetlands on the pre-mined landscape. The final goal is to design a landscape to support and enhance wetland functions at the landscape scale. In some ways, it is an exercise in examining how managers could use ecological functions to meet the criteria for equivalent capability using a scientific approach.

1.2 A landscape scale approach to mine site reclamation planning

While reclaimed areas of Genesee have been deemed successful so far, managers have recognized a need to revise the original reclamation plans in response to changing global and social demands and pressures to integrate long term agricultural and environmentally sustainable development practices. Genesee has conducted on-site biomonitoring, but innovation and research has been limited to site-specific work including ‘live root transfers’ and a ‘mine marsh reclamation project’ (Shifflet 2005). While this approach addresses the question of equivalent capability, it is limited by scale because it is only applicable to ecological features (such as wetlands) as independent elements within the landscape. Though it remains necessary to conduct local scale (100s of m^2) and site-specific (100s of $m^2 > < km^2$) research, a holistic approach at the landscape scale must be considered when evaluating reclamation success, particularly when the impetus to restore ecological processes is driven by the overall reclamation objective (Smyth and Dearden 1998a). As a correlate to site-specific analysis, a landscape scale approach to Genesee’s reclamation planning will help to define optimal land use on a potentially agri-environmental and multi-functional landscape.

1.3 Geographical Information Systems and Spatial Analysis

Spatial analysis should be used to determine optimal land use and Geographical Information Systems (GIS) now make it easier to quantify and qualify entire landscapes at variable spatial and temporal scales. At Genesee, for example, GIS makes it possible to virtually re-create the pre-mined landscape by giving spatial reference to historical aerial photographs. Studying the pre-mined landscape is critical to define the upper and lower limits of site-specific ecological capability and the potential influence these limits will have on wetland restoration (Bedford 1999). In essence, determining equivalent capability for wetland restoration on a disturbed landscape is contingent upon building a landscape scale specific ruler or index to evaluate the post-mined landscape elements. Although many landscape scale studies have been conducted in the past using manual methods (Rogers and Meyers 1980; Auclair 1976; Drew and Shanks 1965), recent advances in software applications such as GIS have facilitated a whole new genre of studies capable of dealing with the much larger file sizes necessitated by spatial data. As a tool, GIS is both multidimensional (beyond the realm of two-dimensional) and dynamic (capable of capturing changes over time) (Lee and Molenaar 2000), and this makes it an excellent candidate for measuring equivalent capability for Genesee at the landscape level. One objective of my study is to demonstrate, virtually, that landscape integrity depends upon the sum of its elements (feature classes) and associated landscape composition (number, type and abundance of features) and configuration (spatial arrangement of features).

1.4 Research question

With disturbed land and access to machines that can reconfigure the landscape, mine managers at Genesee want to know how much of this landscape should be used for wetlands and where it is best to create them. Arguably, wetlands could be put back exactly where they were before mining and this would satisfy the equivalent capability criteria. However, equivalent capability is only assessed with respect to the pre-mined conditions and it is necessary to quantify and evaluate capability before restoring ecological land features and reclaiming productive land uses. Ecological functions depend on the structure of wetlands (Burbridge et al. 1994), and it is important to know if wetlands on the Genesee landscape were optimally arranged to support wetland functionality before it was mined. To qualify the potential of the Genesee landscape to support wetlands prior to mining I re-constructed past conditions using aerial photographs and GIS, quantified the pre-mined landscape using spatial analysis, and used what was measured from the past to provide a reference condition for the future landscape. The overall approach of this research was to establish a reference for equivalent land capability at the landscape scale which compliments past and present biomonitoring efforts at the mine.

1.2 Literature review

1.2.1 Reclamation associated with mining

Wetlands may be impacted by coal surface mining due to "direct removal, acid drainage, sedimentation, and altered hydroperiod" (Mitsch et al. 1983) and therefore, wetland mitigation plans should be initiated and integrated upon mine start-up (Brooks 1990). Moreover, reclamation objective(s) for wetland mitigation associated to surface mining should be site specific, related to regional wetland trends, and free of conflict with other proposed objectives (Brooks 1990). The earlier the integration of wetland mitigation plans, the greater the chance of wetland success qualified by wetland functionality (Brooks 1990). For Genesee, the opportunity to restore wetlands now while the land is in a state of on-going reclamation may prove to be efficient and cost-effective.

1.2.2 Reclamation Plans for Genesee

In 1981, a conceptual reclamation plan for the Genesee Coal Mine was drafted to initiate mining activities in accordance with Alberta Environments' Land Conservation and Reclamation Act (Western Soil and Environmental Services 1981). This plan projects that the mine site will be reclaimed to agricultural land use with intermittent wildlife habitat where biophysical conditions were considered sub-standard for agriculture. Though acceptable at that time, nearly 28 years ago, an extensive body of knowledge has evolved from an emphasis on homogenous land cover to a focus on heterogeneous landscapes that offer greater resilience to disturbances that compromise ecological functions (Franklin 2005). Specifications for land slope and soil depth where forage crops will be seeded are given along with recommendations for re-establishing indigenous plant species in areas slated for wildlife habitat. Water drainage plans consist of channeling water through ditches and outlets into the 'natural drainage system outside the mine area' (such as the cooling pond) and emphasis is placed on draining surface water away from the site and away from agricultural activities. One specific practice outlined in the original reclamation plan is to fill in 'small depressions that may limit agricultural productivity' which demonstrates the focus on altering the natural state of the land rather than capitalizing on the services that could be provided by integrating natural features into the agronomic landscape.

When the Genesee reclamation plan was written (1981), reclamation research was centered on replacing topsoil and re-establishing vegetation and hydrology specifically for agricultural (Western Soil and Environmental Services 1981). Evaluation of reclamation success was limited to measures of soil fertility and moisture, cereal crop yield, and soil erosion. On-going monitoring was recommended for 1) groundwater using test-holes to observe water levels and quality, 2) biology using wildlife studies and, 3) reclamation plots using field designs to test preferred subsoil mixtures. Although the rationale was arguably utilitarian, scientific evidence for restoring ecological integrity was in its infancy, was not yet being incorporated into reclamation plans, and was not deemed necessary.

1.2.3 Environmental Regulation in Alberta

Even though the Surface Reclamation Act (SRA) was passed in Alberta in 1963 (Smyth and Dearden 1998b) in response to increased mining activities, it merely set minimums for mitigating disturbance and re-contouring the landscape. The SRA introduced the concept of 'reclamation certification' and provided the impetus for the following legislation in 1973: the Land Surface Conservation and Reclamation Act (Bratton 1987). This act was the catalyst, requiring reclamation plans for industrial operations in Alberta for all proposed developments. In 1976, A Coal Development Policy for Alberta was introduced to ensure that the primary objective of reclaiming mined land be to restore biotic and abiotic land components to a status similar to that before the disturbance, based on the premise that the reclaimed landscape be utilitarian by design (Province of Alberta 1976). Additionally, this policy introduced the idea that the land could be made "more productive, useful, or desirable than it was in its original state" and insinuated that reclamation efforts should make 'improvement upon [the landscape]' part of the overall goal. Since the conception of the Genesee Coal Mine, the terms reclamation and sustainable development have become more synonymous (Bradshaw 1997), environmental awareness more mainstream, and government and public consultation with industry has become more so the standard. In 1991, Alberta Environment introduced the Environmental Protection and Enhancement Act (EPEA) to facilitate public participation in reclamation activities, particularly those affecting the environment and the potential sustainability of ecosystems long after mine closures (Province of Alberta 1992). The LCRSA guided the formal application for environmental approval to initiate mining operations at the Genesee site but present-day land use planning requires the adoption of current legislation under the EPEA. Further consideration for sustainable landscapes is promoted by Alberta's more recent Land Use Framework which recognizes that wetlands and forests, for example, are public resources that provide ecological goods and services to society as a whole (Government of Alberta 2008). One vision of the framework is to support healthy ecosystems and environment for future generations by maintaining and even enhancing the capacity of ecological goods, such as wetlands, to provide services. This stresses the need to consider what the ecological implications of reclamation, restoration, or rehabilitation will be for the future residents of Genesee and the immediate landscape. Concomitantly, the Provincial Wetland Restoration/Compensation Guide (Alberta Environment, 2007) will facilitate wetland compensation where wetland loss is unavoidable due to development. Together, these policies and guides support the need for science-based research to design self-sustaining and resilient landscapes for the future. A final landscape scale reclamation plan that conforms to these documents will likely maximize the probability of success in building a sustainable landscape including functioning wetlands.

Equivalent Capability at Genesee

For Genesee, reclaiming land to its pre-disturbed conditions may not be a desirable goal because firstly, the geomorphological characteristics of the mined land have been drastically altered and secondly, it would be illogical to assume that the surrounding landscape has not changed throughout the Genesee mine lifespan. In lieu of this, present-day reclamation success is more likely to be a measure of land capability in terms of functional equivalence rather than equality. It may be difficult to restore or create wetlands that provide all the functions at the same level as reference wetlands, thus functional equivalence allows for an approximation for functions such as nutrient cycling (Urbanska et al. 1997). This approach is more realistic in areas like Genesee where ecological values have evolved along with a global demand for sustainable ecosystems since the mine was initiated over 25 years ago. McQueen et al. 1991, used the Highvale Coal Mine in Alberta (adjacent to the Genesee Mine) to illustrate the importance of assessing pre-mine land capability to predict post-mining capability and to incorporate appropriate reclamation procedures during mining (as opposed to after mining) to meet reclamation goals. McQueen et al.'s study demonstrated that the sooner landscape capability was incorporated into the reclamation planning process, the more efficient the process of achieving reclamation goals were. Equivalent capability accommodates the reality that land uses and land cover may change in the course of operations as a result of natural forces (e.g. changing climate, hydrological regime) or anthropogenic requirements (e.g. alternative food sources, ecological reserves, natural resource preservation). At Genesee, land capability has historically been associated with agronomic land classes, but there is a growing need to integrate natural land classes into the landscape. Knowing the range of options available for reclamation allows for evaluation with a spatial model of the landscape. This modeling approach will allow *a priori* decision-making on landscape configuration, water body placement, and ultimately, optimal LULC upon mine closure.

1.2.4 Importance of wetlands

As our human population increases we require more land for agriculture, resource access, dwellings, factories, and recreation. In Canada, the total number of farms has decreased while the total land area covered by the remaining farms has increased (Lefebvre et al. 2005). Throughout North America, this trend of land conversion to agriculture has replaced millions of kilometers squared of wetlands (Ramankutty and Foley, 1999). Ecologically, wetlands are important to biodiversity (Corley et al. 2006; Kirkman and Golladay 1999), wildlife habitat (Amezaga et al. 2002; Reunanen et al. 2000), and water quality (Jobbagy and Jackson, 2004; Mitsch and Gosselink, 2000). These are only a few examples of the services that can be provided by wetlands but they are important, and arguably irreplaceable. Accordingly, it has become increasingly important to quantify what remains on the landscape so that baselines or reference states may be established. The configuration (spatial arrangement) and composition (presence and amount of land cover types) of these landscape features is indicative of the processes that may have created the patterns and indicative of ecological mechanisms (Griffith 2002).

Wetlands and riparian zones

Associated with wetlands are the riparian zones; the transitional zone from water's edge to upland areas; a water-affected community that is distinct from its surroundings (Mitsch and Gosselink 2000). Riparian zones are ecologically important to wetland biota (e.g., invertebrate diversity/abundance, vegetation biomass; Batzer et al.

2000 and abiota (e.g., biogeochemical processes and sedimentation; Clement et al. 2003, Lockaby et al. 2005), and riparian zones are economically important for wetland carbon sequestration (McCarty and Ritchie 2002). As "complex ecological systems", riparian zones are interrelated with the physical characteristics and the geomorphological processes associated with the proximate wetlands (Naiman et al. 2000). Many functions pertinent to wildlife, material, and nutrient transport have been identified and related to riparian buffer structure (composition and configuration).

Wetlands and spatial attributes

Wetland size (area) and shape (area to perimeter ratio) matters for 1) microorganisms (e.g. algae), 2) invertebrates (e.g. aquatic insects), 3) plants (e.g. macrophytes), 4) animals (e.g. birds, fish, amphibians, mammals) 5) wetland physical and chemical properties, and 6) hydrological function (Van der Valk 2006). Nekton and benthic infauna, for example, are most productive in wetland habitats where 'edge' habitat is maximized and spatial analysis has been used to monitor restored wetlands exclusively for this edge quality (Feagin and Wu 2006). Anuran species abundance is affected by surrounding land uses (e.g. forest, urban, agriculture), proximity to other wetlands, and anthropogenic features (e.g. roads), and the distribution pattern of anuran abundance relative to landscape features has been demonstrated using spatial analysis (Gagne and Fahrig 2007, Knutson et al. 1999). Aznar et al. (2003) used 'density' and 'connectivity' as landscape metric indicators to measure wetland patterns caused by 'human pressure' to predict vegetation assemblages and the associated biological state of wetlands. On a broader scale, wetland composition and configuration affect, and are affected by, LULC change and quantifying wetland structure at the landscape scale makes it possible to detect trends of wetland quality and project future management implications (Torbick et al. 2006).

Wetlands and roads

Roads and frogs do not mix well. More specifically, it is the vehicles moving on the roads that have negative implications for most anurans (Eigenbrod et al. 2008; Hels and Buchwald 2001; Pellet et al. 2004). Since many anurans are dependent upon wetlands for habitat, food, and breeding (Semlitsch and Bodie 2003), wetland distances from roads are important for anuran survival. Conversely, amphibians are beneficial because as predators and prey, they influence the biotic and abiotic characteristics of wetlands (Klaus and Azous 2001). Some of the anurans that may be found at Genesee include the northern leopard frog (*Rana pipiens*), the boreal chorus frog (*Pseudacris maculata*), the wood frog (*Rana sylvatica*) and the western toad (*Bufo boreas*)

Unlike frogs, non-native plant species and roads do mix well. Roadside ditches are designed to drain water away from roads but in doing so, ditches provide the perfect hydrological highway for aquatic invaders such as *Phragmites australis*, to travel along (Lelong et al. 2007; Maheu-Giroux and de Blois 2007). Once established, non-native wetland plant species are difficult to control and can negatively affect native species (Byers et al. 2002). Overall biodiversity of native wetland plants and animals have a negative correlation with increasing road density (Findley and Houlahan 1997) and furthermore, the effects of roads on wetland biodiversity are cumulative over time (Findlay and Bourdages 1999). This effect of roads on the ecological system has been dubbed the "road-effect zone" by Forman and Deblinger (1999) and refers to the area adjacent to a road that it is affected by the presence and usage of the road. After measuring nine ecological factors, one of which was wetlands, Forman and Deblinger (1999) identified road-effects up to and beyond 100m from road-edge for all factors and deduced 600 m as the average road-effect distance. The implications of road density for wetland health at the landscape level are overwhelmingly negative.

Because they are impervious structures, the construction, use, and maintenance of roads affect both the physical environment (soil density, temperature, soil water content, light, dust, surface-water flow, pattern of run-off and sedimentation) and the chemical environment (heavy metal contaminants) (Azous and Horner 2001 (pp.66); Trombulak and Frissell 2000). Biotically (e.g. road kill and infestation) and abiotically (physical and chemical), wetlands are affected by proximity to roads (Donaldson and Bennet 2004). Previous research suggests that vehicle emissions decrease exponentially with distance from roadway (Kirchner et al. 2005) and in the extreme case of a Los Angeles freeway, the zone of greatest influence was within the first 500 m (Rodes and Holland 1981). The distance in which road effects are most likely to be observed are within 300 meters and as far as 2000 meters away from a road surface (Houlahan and Findlay 2004).

1.3 Overview of thesis chapters

Excavation for open-pit mines disrupts vast areas of land, alters local topography and hydrological pathways (Younger et al. 2002), and invariably influences the abundance and spatial distribution of wetlands on the landscape. Coal mining mitigation offers the potential to restore or possibly even create wetlands. Wetlands are important because they may provide immeasurable goods and services to humans, such as habitat provision, nutrient and material transport, water quality enhancement, ground water recharge, and even carbon sequestration. The use of historical conditions as a reference is one approach to restoring natural wetland conditions. A landscape scale approach with GIS and spatial analysis may increase understanding of historical landscapes in the absence of pre-disturbed ecological data and improve the reclamation potential to enhance the provision of wetland goods and services on a post-mined landscape.

In chapter two I examined the relationship between wetland presence and landscape features which were categorized as either anthropogenic or natural. Historical aerial photographs (1982), GIS, spatial analysis, and a model were used to develop surrogate measures of wetland presence on a pre-disturbed landscape. The distribution of wetlands for the whole study area as well as for two subset categories: agricultural land and non-agricultural land were documented. Due to the small land area (~7400ha) allocated for the mine permit, the surrounding watershed area (for a total of ~28, 444 ha) was included to increase the power of observed relationships between wetlands and landscape features on the pre-mined site. For chapter two, the main objective was to use GIS to explain why wetlands occurred where they were on the landscape prior to mining. This was divided into two sub-objectives: First, to examine the landscape factors that explain the presence of wetlands on agricultural lands versus non-agricultural lands and second, to develop a model to explain the relationship between wetland presence and landscape factors.

In chapter three, I compared the pre-mined landscape (1982) to the present-day mine site (2007) using the spatial extents of the present day mine permit for both time periods. The objective of chapter three was to investigate wetland composition, configuration, and overall landscape structure change between 1982 (pre-mined) and 2007. This was divided into three sub-objectives: First, determine which landscape metrics best represent the state of wetland integrity. Second, identify spatial variability of wetlands and explain differences between the two time periods. Third, measure landscape heterogeneity and explain differences between the two years (1982 and 2007).

Chapter four provides a summary of chapters one and two, the implications that the study results may have for wetland restoration, the limitations of this research and some recommendations for future work based on the results of this research.

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CHAPTER 2 Explaining wetland position

2.1 Introduction

Wetlands are valued by humans for the many functions that they perform and the services they provide, and this value is site-specific (Shiratani et al. 2006, Woodward and Wui 2001). Wetlands provide 1) habitat for aquatic invertebrates and nearly 600 wildlife species in Canada 2) pollution control via phosphorus burial and denitrification (conversion of nitrates/nitrite to gaseous nitrogen) and 3) soil erosion control (Ducks Unlimited Canada 2004). These are just a few examples of inherent wetland processes valued by humans that justify the need to protect or restore wetlands. Theoretically, wetland values will be enhanced when they persist where the hydrogeomophic potential of the landscape is optimized for supporting wetlands (Whittecar and Daniels 1999, Bedford 1996). A landscape is described by Forman and Godron (1986) as a heterogeneous land area composed of a cluster of interacting ecosystems that is repeated in similar form throughout. At the landscape scale, there exists an interactive and dynamic relationship between spatial pattern (arrangement of landscape elements) and ecological characteristic (Risser et al. 1984), and for wetlands, the implication is that physical landscape form influences wetland function.

Determining which landscape features best explain the presence of wetlands prior to disturbance is essential when the objective of restoration is to replicate or enhance the potential for wetland functionality. Where a wetland is positioned on the land and the topography associated with that position set many of the limits and conditions on biophysical processes affecting wetland character (Mitsch 1992). The spatial location and the relative land position of a wetland determines which wetland services can be provided and emphasizes how the perceived benefits of these services influence the goal of wetland restoration (Rapport et al. 1998). Wetlands occurring above a pollution source, for example, can do far less to mediate the water-borne pollutants than a wetland situated below the location where pollutants are introduced. Wetland juxtaposition (e.g. embedded in an agricultural matrix, occurring on mine tailings, within a network of channelized inflows, or in a porous groundwater recharge zone) has a large bearing on the character, appearance and function of wetlands. Wetland benefits are influenced by topography (slope and elevation) and adjacent land cover and use. Examples of these relationships from scientific literature are illustrated in Table 2.1.

Under the Provincial Wetland Restoration/Compensation Guide (Alberta Environment 2007), “Wetland compensation in Alberta is a process that requires the restoration of drained or altered naturally occurring wetlands”. However, when a wetland is drained away from its original location for agriculture or displaced for the development of roads and farmsteads, the processes explaining its presence may be described as anthropogenic rather than natural. Accordingly, if a repositioned wetland is found on land considered sub-optimal for wetland function, should it be left in that location even when there is opportunity to re-design the landscape and optimize wetland placement? The Genesee mine site provided an opportunity to examine the status of wetlands on the pre-mined landscape before reclamation. Understanding which land forms, uses, or covers explained wetland position prior to disturbance helps to evaluate wetland position and potential functions provided prior to disturbance and aid wetland restoration by providing a reference for equivalent land capability. In the context of this study, restoration refers to the process of bringing back natural ecosystems (e.g. wetlands) to a prior state with the intent being “reinitiating ecosystem processes” whereas reclamation is used to refer to the process of returning land to a former production system (e.g. agriculture) (Clewell and Aronson 2007). Reclamation is the term more commonly used when referring to overall land use practices and post-mining activities at Genesee.

2.1.1 Research Hypothesis

Agricultural land use influences the spatial distribution and abundance of wetlands. I hypothesized that pre-mined wetland position on the landscape would be better explained by adjacent land uses which are mainly agronomically-related, than by land form. The main premise of this hypothesis is that wetlands would have been drained away from prime agricultural land into areas classified as less productive or near roadways, a common practice during settlement. If wetland position is explained by topographic and natural land features, like slope and proximity to tributaries, this could suggest that the pre-disturbed wetland structure approximates a natural state. Conversely, if wetland position on the pre-mined landscape is determined mainly by land use and anthropogenic land covers, such as crops and roadways, this could suggest that the wetland structure approximates a previously disturbed or unnatural state. As implied earlier, this distinction between a natural and unnatural state, where state is defined as a "manifestation of an ecosystem in terms of how it appears" (Clewell and Aronson 2007), is relevant to defining landscape form and structure and evaluating the potential relationship with wetland functionality.

2.1.2 Research Objectives

The main objective of this research was to virtually replicate the landscape using Geographic Information Systems (herein GIS) to develop explanations for why wetlands occurred where they were on the landscape prior to mining. More specifically I wanted to:

- Identify and map wetland structure in the mine permit and surrounding land area using pre-mined (1982) aerial photographs.

- Identify and map all Land Use and Land Cover (LULC) as well as physical land form characteristics in the mine permit and surrounding land area using pre-mined (1982) aerial photographs and a Digital Elevation Model (DEM).

- Create a descriptive model to explain the historical pattern of wetlands on the pre-mined landscape to make the relationship between wetland presence and landscape factors explicit.

2.1.3 Research Benefits

Results from this study could augment land use planning efforts by reducing the number of sites considered capable of supporting and sustaining wetlands. The final model is intended to be used as a tool to evaluate historical reference systems and facilitate future planning at the landscape scale. With an objective approach, my research was designed to explain wetland position on the pre-mined landscape and increase the likelihood of establishing resilient and sustaining wetlands on a post-mined landscape. At a broader scale, this research demonstrates that each restoration project is unique and therefore requires a comprehensive and individual site assessment to facilitate successful restoration (Clewell et al. 2007).

2.2 Study Site

The Genesee Generating Station, owned by Capital Power (formerly Epcor) uses coal-fired generators to provide electrical power to Edmonton and the surrounding communities. Coal is provided by an adjacent strip-mine, jointly owned (50/50) by Capital Power and Sherritt Coal. The mine site is located approximately 70km south-west of Edmonton, Alberta (53^{o} 20’ 35” N, 114^{o} 18’ 17” W) in the North Saskatchewan River Basin (Figure 2.1). The study area falls within the Edmonton Plain sub region of the Eastern Alberta Plains (Pettapiece 1986) and is considered a Dry Mixedwood Sub region of the Boreal Forest Natural Region (Achuff 1994). The landscape is characterized mainly by undulating lowlands and hummocky uplands with clay-textured parent materials, luvisolic and organic soils and an agro-climate of moderate heat limitation (Pettapeice 1986). Summers are short and warm (July average 16.3°C) and winters are long and cold (January average -13.4°C) (Westworth and Brusnyk 1983). For the mine permit area only, local topography and landscape were described by Western Soil and Environmental Services (1981) prior to mining as

> The region is underlain by a succession of essentially terrigenous sandstones and shales, and coal beds of late Cretaceous and early Tertiary ages. These have been very slightly deformed by regional tectonic and local glacial processes.
>
> Present day uplands reflect preglacial highs to some extent but glacial and alluvial deposits generally subdue bedrock relief. In the Genesee area, the topography varies from gently rolling to flat and slopes generally in a northeasterly to northwesterly direction. Topography ranges from 782m to 730m.

At Genesee, The “upper Ardley coal zone of the Cretaceous-Paleocene Scollard formation is mined for use as thermal coal” (Pollock et al. 2000). The upper Cretaceous and Paleocene stratigraphy consists of the the Paskapoo Formation, Scollard Formation, the Battle Formation, the Whitemud Formation and the Horseshoe Canyon Formation (Pollock et al. 2000).

2.3 Materials and methods

Data overview

I used ArcGIS desktop software (version 9.2) along with the Spatial Analyst extension (Environmental Systems Research Institute Inc [ESRI] 2006). All the data used for this study were projected using a Universal Transverse Mercator Projection (UTM) on the North American Datum of 1983 (NAD83) to match a current (2007) orthoimage (1:30,000) that was georeferenced using ground control points (GCP) from the Genesee mine-site. I used two main data sources from which all the feature classes were derived; aerial photographs and a Digital Elevation Model (DEM). I used a 1 x 1 m resolution for all raster analysis. I used a 1 x 1 m resolution for all raster analysis. Resolution refers to how accurately the map features can be depicted at a given scale and map scale is the extent of reduction expressed as a ratio (e.g. 1:10,000).

2.3.1 Data preparation

The response variable was binomial; either a wetland was present (1) or not present (0) and the predictor variables included 1) slope, 2) elevation, 3) sinks, 4) compound topographic index (CTI), 5) topographic ruggedness index (TRI), 6) distance to roads, 7) land use (agricultural or non-agricultural), and 8) distance to rivers or streams. Predictor variables are explained below.

2.3.1.1 Source data

Aerial photographs

I obtained pre-mined (1982) black and white aerial photographs (1:10,000) from EPCOR. The photographs were scanned on a flatbed scanner using a resolution of 600 dots per inch (dpi), georeferenced, rectified and mosaiced to create four consecutive map sheets. Photographs that did not overlap with the 2007 orthoimage were georeferenced using 20K base features (scale 1:20,000) obtained from AltaLIS, a not-for-profit agent for Spatial Data Warehouse Ltd, (SDW) that maintains Alberta’s digital mapping. Exact registration between all photographs was not possible and had to be corrected for during manual delineation and classification of landscape features.

Digital Elevation Model

The Digital Elevation Model (DEM) was created from a point shape file provided by AltaLIS (obtained through the University of Alberta's spatial data library). Source data for the point file was gathered prior to 1984 (AltaLIS) making it suitable for this pre-mine (1982) analysis. From the point shape file I was able to create a triangulated irregular network (TIN) which was then converted to a raster format using a one meter (1 m) cell size.

2.3.1.2 Land form data

Wetlands may be described or defined in great detail but essentially all definitions may be summarized by three consistent characteristics: shallow water, hydric soil and adapted plants (Tiner 1999). One of the strongest controls on these characteristics that define a wetland are geomorphic setting (Brinson 1993) and "one of the strongest controls on the water balance of a wetland is topography" (Richardson et al. 2001). Accordingly, I created five raster feature classes to capture landscape form affecting surface hydrology: 1) slope, 2) elevation, 3) sinks 4) compound topographic index (CTI), and 5) topographic ruggedness index (TRI). Each of these is described below. Each of the five feature classes derived from the DEM are shown in Figure 2.2.

Slope

Wetlands are generally found on flatter terrain and are less likely to be found on steeper slopes (Mitsch and Gosselink 2000), and this makes slope a good geomorphic indicator for wetland position. Slope can be used to predict where wetlands may occur on a landscape because accumulation of water at specific points on a landscape is influenced by slope gradient and slope length (Richardson et al. 2001). Slope is calculated by the difference in elevation values from cell to cell on a DEM and I was able to create a slope feature class from the DEM using 3D Analyst in ArcMAP 9.2 (ESRI 2006) using percent for an output measurement.

Elevation

Wetlands are characterized by hydric soils which are "soils that are saturated, flooded, or ponded long enough during the growing season to develop anaerobic conditions in the upper part" (Soil Survey Division Staff 1993). Hydric soils, and consequently wetlands, often occur in areas of lower elevation where surface water inflow is likely to be greater than outflow. Without specific data for groundwater, it is difficult to predict where wetlands will be affected by subsurface water flow on a relatively flat landscape based on elevational differences alone. Elevation values for the study landscape vary between 649 m and 833 m with a mean elevation of 741 m and a standard deviation of 37 m; extreme values are due to ravines. To emphasize the difference between the middle values, I reclassified the elevation raster into ten classes using a quantile method where values are divided equally among classes. The elevation feature class was created from the DEM using spatial analyst extension (ESRI 2006).

Sinks

Sinks occur where flow in is greater than flow out, as areas of "internal drainage", or in a virtual sense, where a raster cell or group of cells is surrounded by cells with higher elevation values (ESRI 2006). Sinks are low-spots on the landscape where water is likely to persist and consequently, all sinks on the raster were reclassified with a value of one. An integer raster for the sink feature class was calculated using the sink tool in the spatial analyst extension (ESRI 2006).

Compound Topographic Index (CTI)

Hydric soils may be organic or mineral and other than the percentage of carbon, these soils differ in 1) bulk density and porosity, 2) hydraulic conductivity, 3) nutrient availability, and 4) cation exchange capacity (Mitsch and Gosselink 2007). These soil properties determine the wetland environment and influence the potential for a wetland to perform functions such as denitrification which is important in an agricultural landscape (Groffman and Hanson 1997). Soil data for Genesee and the surrounding landscape was not available at an adequate scale (with enough detailed soil information) and therefore the DEM was used to derive values known to be related to soil characteristics. The Compound Topographic Index (CTI), or wetness index, uses upslope catchment area and slope from a DEM to calculate the saturation potential for each cell (Moore et al. 1993). The CTI quantifies catenary landscape position with an index where lower values represent upper catenary positions and higher values represent lower catenary positions (Gessler et al. 2000). The CTI is highly correlated with soil attributes (horizon depth, silt percentage, organic matter content) (Moore et al. 1993) and is commonly used to capture the spatial distribution of soil properties as a function of topography (Yang et al. 2007; Gorsevski et al. 2006; Ziadat 2005; McKenzie and Ryan 1999). I created a CTI grid using ArcMap 9.2 and a CTI ArcScript from Evans (2004) shown as:

CTI = ln (A_s / (tan(beta))

Where A_s = Area Value calculated as (flow accumulation + 1) *(pixel area m2) and beta is the slope expressed in radians.

Topographic ruggedness index

Wetland hydroperiod (seasonal flooding pattern) is influenced by terrain (Mitsch and Gosselink 2007) and more specifically, it is the movement of water over and through the landscape that is influenced by topography (slope and elevation) (Richardson et al. 2001). A rugged terrain is more likely to have "poorly integrated surface drainage" compared to a relatively smooth terrain where movement of subsurface water is simplified (Richardson et al. 2001). The Topographic Ruggedness Index (TRI) uses a DEM to calculate the difference in elevation for each cell based on the eight neighboring cell elevation values to attain an overall ruggedness index for the landscape (Riley et al. 1999). I created a TRI grid using ArcMap 9.2 and a TRI ArcScript from Evans (2004) shown as: TRI [DEM] [OUTGRID] {CLASSIFY}.

2.3.1.3 Land use and land cover (LULC) data

To create LULC feature classes, I used editor (sketch tool) (ArcMap 9.2, ESRI 2006) to manually digitize all land classes that were visible and identifiable on the orthoimages at a viewing scale of 1:2000 for the entire landscape. Feature classes, including agricultural land, structures, roads, rivers and streams and tributaries are shown in Figure 2.3 and described in Table 2.2. Ground-truthing was limited to landscape features that remained relatively unchanged from 1982 to 2007, but I was able to reference relative land classifications using a classified image created in 1998 for biomonitoring purposes by Jacques Whitford and Associates (a consulting firm). AltaLIS base features, including roads and water bodies, were overlaid and used for relative placement and classification but were generally too coarse to be of any direct use. All data feature classes were clipped to the extents of the defined study area and all analyses were conducted using a mask setting in ArcToolBox (ArcMap 9.2, ESRI 2006).

Distance feature classes were created for both the road and hydrology (lotic) features where potential effects to wetland function were expected to depend on the distance from that feature. Road-effects were considered negative and were expected to decline with increasing distance-to-wetland. Conversely, the proximity of a wetland to other hydrological features was considered beneficial (positive) and effects were expected to decline with increasing distance-to-wetland. All distance variables were measured in meters using euclidean distance. Road effects were expected to decline exponentially with distance rather than linearly thus surfaced roads (rds) and non-surfaced roads (rdg) were transformed to exponential decays of the form $w = e^{-p*d}$, where w is the weight of the function, e is the exponent function, $-p$ is the decay parameter, and d is the distance from road value (Nielsen et al. 2009). The negative value for p represents the rate of road-effect attenuation at greater distances from the road. The difference between a power function and an exponential decay function is that 1) the power function will return a value of infinity when distance is equal to zero (d=0) whereas the exponential decay function will return a value of 1 and 2) the power function will return higher values for nearer values than will the exponential function (Wise 2002). For my analysis, the decay function was more appropriate than a power function to represent road-effects on wetlands.

The exponential decay ranged from a value of 1 at the road to 0 at distances far from the road. In raster calculator I calculated the exponential decay function for each road feature using: [NED_M] = Exp([roads_dist] * -0.01)

Where *NED_M* is the negative exponential decay value for surfaced roads, roads_dist is the Euclidean distance raster for main roads, and -0.01 is the decay parameter. To determine the value of p (decay parameter), I evaluated the response of five potential values (0.1, 0.05, 0.02, 0.01, 0.005) (Figure 2.4).

Using a decay function equal to -0.01, a weight of 1 is given at the road and a weight of approximately 0.05 at a distance of 300 m. A decay value of -0.01 represents a subjective approximation of road-effect distance whereby most (95%) of the effects of roads on wetlands will occur at distances of 300 m or less from the road. At distances of greater than 300 m, the decay function continues from 0.05 to infinity.

2.3.1.4 Wetland (response) data

Editor (sketch tool) in ArcGIS 9.2 (ESRI 2006) was used to manually digitize all wetlands that were visible and identifiable on the orthoimages at a viewing scale of 1:2000 for the entire landscape. Wetlands were classified as one of four types: saturated soil, beaver pond, dugout, or natural wetland. The classification of saturated soil was used for land that appeared distinguishably wetter than surrounding land; characterized by dark coloured soil and less vegetation. Beaver ponds were distinguished by impounded areas of water along a tributary – often preceded by the hard break line indicating a beaver dam. Dugouts were easy to distinguish as rectangular and vegetation-free water bodies near, or surrounded by, agricultural land use. The classification of natural wetland was reserved for all distinct entities of standing water not falling into the preceding three categories. Wetland boundaries were considered to be water's edge, or to the best interpretation of where plant cover appeared greater than open water. Keeping with the objective of restoring wetlands to the post-mined landscape, only the wetlands classified as natural were used for this analysis.

2.3.2 Model preparation

I used the 'feature to point tool' in the Data Management toolbox (ArcGIS 9.2. ESRI 2006) to create a point feature class from the wetland polygon feature class. Each output point, representing the centroid of a wetland, was assigned a value of 1 for wetland presence. For the random points, I generated one thousand points using the program Hawthstools Ver.3.27 (Beyer 2006) in ArcGIS 9.2 (ESRI 2006). Each random point was assigned a value of 0 for wetland absence. To avoid having random wetlands on top of, or too close to existing wetlands, I first set the minimum between points distance to 17 m (derived from analysis of existing wetlands) and then prevented points from landing within polygons on the original wetland feature class. Both true wetland sites (1) and random sites (0) were used to extract values (attributes) from each predictor variable feature class. The analysis-ready dataset consisted of 265 wetland points and 1000 random points for a total of 1265 points attributed with values from each candidate covariate (Figure 2.5).

I used two approaches to modeling the Genesee landscape. For the first approach, I included all the data in one model and included an interaction term for agricultural. The interaction term was added to assess whether there was a difference in the response variable (wetland presence or absence) when land use was equal to either agriculture (1) or non-agriculture (0). For the second approach, I separated the data into two data subsets using the agriculture designation (0,1) and generated a separate model for each subset to calculate simple effects not captured in the comprehensive model with the interaction term. Simple effects are the effect of an independent variable at a single level of another variable.

I used eleven continuous covariates: 1) elevation (elv), 2) slope (slp), 3) soil wetness (CTI), 4) terrain ruggedness (TRI), 5) distance from surfaced roads (rds) 6) distance from non-surfaced roads (rdg), 7) distance from lease roads (rdl), 8) distance from private roads (rdp) roads, 9) distance from tributaries (trb), 10) distance from river/stream (riv) and 11) distance from structures (str). I used two categorical covariates: 1) land cover identified as agricultural or non-agricultural (agr), and 2) sinks (snk). Feature class details are shown in Table 2.3.

Statistical Analysis

All analyses were conducted using the statistical software package R, Ver. 2.5.1. (R Development Core Team 2007). Exploratory analyses were conducted to determine the structure of the relationship between wetland presence and each covariate. I used a univariate generalized linear model (GLM) (Hosmer and Lemeshow 1989) to investigate the probability of wetland presence based on a vector of explanatory variables and interaction terms of particular interest, mainly the interaction between the agricultural land class and each other covariate. Covariates were assessed and ranked by residual deviance prior to being included in the full model. Residual deviance is “twice the difference between the maximum achievable log-likelihood and that attained under the fitted model” (McCullagh and Nelder 1989); used to measure a model’s goodness-of-fit. Each covariate was inspected visually using bar plots for categorical data (agricultural land use and sinks) and boxplots for all other data (elevation, slope, soil wetness, terrain ruggedness, distance to roads, distance to rivers/streams, and distance to structure). To assess collinearity among explanatory variables, I used Pearson’s correlation ($|r|$). When correlation between two variables was high ($|r| > 0.6$), I removed the variable considered to be less biologically relevant to the presence of wetlands. Explanatory variables were transformed using log (x), square-root (x), and a quadratic (x^2) and inspected for fit before being added to the model. (Hosmer and Lemeshow, 1989).

To measure the probability of wetland presence, I used a GLM with a binomial-response (wetland = 1, random point =0) for the outcome variable and a logit-link function (Hosmer and Lemeshow, 1989). Random selection was used to obtain training data representing 75% of the available wetland and random data points. The remaining 25% subset of data was withheld as testing data for model testing (Hosmer and Lemeshow 1989). Variables for the final model were selected in turn using a forward stepwise procedure (Hosmer and Lemeshow, 1989); beginning with the variable with the lowest residual deviance by univariate analysis, and ending with the least significant variable. In this approach, each term is added to the null model, one at a time, until an included variable is no longer significant at $\alpha = 0.01$. In 'R', the first factor level is automatically treated as a baseline or reference level against which subsequent levels are compared (Maindonald and Braun 2007). To ascertain that remaining and unused variables did not contribute significance to explaining the presence of wetlands, I added each term independently to the reduced model. After all the main effects had been assessed, I added an interaction term to the model to determine if there was a difference between data points associated with agricultural land (1) versus non-agricultural land (0). If there were two variables that could be interchanged but not used within the same model, I created two models and selected the one with the lowest Akaike Information criterion (AIC) weight (Burnham and Anderson 2002) that was also the most parsimonious model. Akaike weights are "the relative likelihood of the model, given the data" (Burnham and Anderson 2002).

To assess the fit of my final model, I used a Receiver Operating Characteristic (ROC) curve (Hosmer and Lemeshow, 2000) with my testing data (25% of data withheld from the analysis). The ROC "plots the probability of detecting true signal (sensitivity) and false signal (1 – specificity) for an entire range of possible cutpoints" (Hosmer and Lemeshow, 2000). The Area Under the Curve (AUC) "provides a measure of the model's ability to discriminate between those subjects who experience the outcome of interest versus those who do not" (Hosmer and Lemeshow, 2000). An AUC value equal to 0.5 would suggest that the model could not predict any better than flipping a coin whereas a value greater than 0.7 and less than 0.8 would suggest an acceptable model, 0.8 to 0.9 an excellent model, and greater than 0.9 – outstanding , though "extremely unusual" (Hosmer and Lemeshow, 2000). To find the optimal cutpoint, I plotted all the values for the sensitivity and specificity curves to determine where they crossed, which is where both sensitivity and specificity are maximized (Hosmer and Lemeshow, 2000).

From the logistic regression model output, the coefficient estimates from the model were multiplied by the covariate grid cell values in raster calculator in ArcMap 9.2 (ESRI 2006) to generate a raster image predicting wetland presence for the entire study area. Raster images represent geographic features as squares arranged in rows and columns wherein each square (or cell) holds a value representative of the underlying data. From the prediction raster, I used probability, P, determined by $P = 1 / (1 + e^{-L})$ to map the probability of a wetland occurring at each site (cell). Probability of wetland presence is based on the covariates used in the model to describe the linear predictor (Nielsen et al. 2007). To predict what reference conditions may have been before anthropogenic influence, I removed model variables categorized as anthropogenic (agricultural land and roads) and applied only the remaining variable coefficients to the probability raster. I applied the cut point (determined from the specificity and sensitivity curves from the model) to the prediction raster by reclassifying the raster to show color coded values for sites predicted to be wetlands by the model.

2.4 Results

2.4.1 Wetland distribution

The Genesee study site was categorized into agricultural and non-agricultural land cover types to allow contrasts. Agricultural land cover represented 18,421 hectares (65 %) of the total landscape. Non-Agriculture land was comprised of seven sub-land cover classes: 1) sand/gravel, 2) coniferous forest, 3) deciduous forest, 4) grassland, 5) mixed forest, 6) shrub and/or willow, and 7) windbreaks; which together represented 10022 hectares (35 %) of the total landscape (Table 2.4). Of the 265 true wetland points, 105 occurred on agricultural land and the remaining 160 (60%) occurred on non-agricultural land. Of the 1000 random points, 655 occurred on agricultural land and the remaining 345 (34.5 %) occurred on non-agricultural land (Table 2.5). The percentage of random points for each land class was representative of the landscape ratio of agricultural land to non-agricultural land.

2.4.2 Factors affecting wetland presence

2.4.2.1 Preliminary assessment of candidate variables

Bar plots and box plots for the study site show graphically the variation of wetland presence rates for random sites (0) versus true wetland sites (1) for each covariate. The bar plots for the percentage of wetlands on agricultural land suggest that there is a difference between random and wetland points. Visual inspection of the box plots suggests that there is a difference between mean value of wetlands (1) and random points (0) for six covariates (trb, slp tri, cti, riv, rds). Differences between the remaining covariates were less noticeable.

The estimated or "fitted values" for each covariate are graphically represented in Figure 2.6. The probability of wetland presence increased with 1) increasing elevation (m), 2) increasing soil wetness (index), and 3) increasing (negative) exponential decay rate to surfaced roads. The probability of wetland presence decreased with 1) increasing (negative) exponential) decay rate to non-surfaced roads, 2) increasing distance (m) to lease, and private roads 3) increasing distance (m) to structures, 4) increasing distance (m) to tributaries, 5) increasing slope (degrees), and 6) increasing terrain ruggedness (index). Lastly, the probability of presence of wetlands on non-agricultural land is greater than on agricultural land.

2.4.2.2 Assessing multicollinearity

The Pearson's correlation coefficient between distance-to-rivers (riv) and elevation (elv) ($r = 0.824$) showed a strong correlation and because elevation differences were due mainly to the river valley (+/- 184m) and considered too trivial to provide discriminatory power, this variable was excluded from the modeling process. Structure (str) and distance-to-private roads (rdp) were inherently and moderately correlated ($r = 0.651$); structure (str) was excluded from further analysis. Slope (slp) was highly correlated with Terrain ruggedness (tri) ($r = 0.974$) and to a lesser extent, negatively with soil wetness (cti) ($r = -0.502$). Soil wetness (cti) and terrain ruggedness (tri) were also moderately and negatively correlated ($r = -0.536$). After examining the covariates using scatter plots (Fig. 2.7), both covariates (tri and cti) were included because a linear relationship was not obvious and both showed significance ($p<0.001$) and low residual deviance values for the univariate analysis. The covariate for slope (slp) was excluded based on multicollinearity. Correlation between any of the remaining variables did not exceed (+/-) 0.475. A scatter plot matrix of the remaining variables demonstrating a weak correlation is shown in Figure 2.8 .

2.4.2.3 Transformations

Each predictor variable was plotted with no transformation, a logarithmic transformation (log), a square-root transformation (sqrt), and a power transformation (quadratic term) ($x+x^2$) to check for linearity and determine if transformations improved the symmetry of distribution. The compound topographic index (cti) did not require a transformation, whereas the terrain ruggedness index (tri) was improved by the logarithmic transformation (log y+1). Variables representing roads (surfaced, non-surfaced, lease) were moderately improved by a square root transformation. Variables representing rivers and streams were improved with the quadratic equation.

2.4.2.4 Training and testing data

A Generalized Linear Model (GLM in R) was fit to the training data (75% random data) with two categorical (snk, agr) and eleven continuous (elv, cti, riv, rdl, rdg, rdp, rds, slp, str, tri, trb) standardized variables to identify the significant predictors and develop a model. Residual deviance values, obtained from univariate analysis, ranged from 952.63 (agr) to 985.55 (snk) (Table 2.6). Land use as either agriculture or non-agriculture (agr) was the most discriminating variable with the lowest residual deviance and was therefore selected as the first step of the stepwise procedure. The fit model included five terms: agriculture (agr), distance to tributaries (trb), terrain ruggedness index (tri), distance to river (riv) and surfaced roads (rds), plus one transformed term (sqrt(trb)) and one interaction term (agr:rds). Residual deviance for the final model for the training dataset was 856.73 (*df* = 942, AIC score = 872.72). Training model results are given in Table 2.7. The remaining test data (25% random sample) was used to assess the model. Residual deviance for the final model for the testing dataset was 279.36 (*df* = 307, AIC score = 295.36). Results from the training model are given in Table 2.8.

2.4.2.5 Receiver operator curve (ROC) and Area Under Curve (AUC)

From the receiver operator curve, the AUC value (0.769) suggested that the model derived from the testing set was ‘appropriate’ for explaining wetland presence

(Fig. 2.9). With the training data, the AUC value was slightly less (0.725) (Fig. 2.10). The difference in accuracy between the training data and the test data was minimal suggesting a robust model.

2.4.2.6 Full data model

When the GLM from the training data was applied to the full dataset (random points (0) = 1000, wetland points (1) = 265), all variables (including the interaction term and quadratic term) were significant at $\alpha < 0.001$, except for rds where $\alpha > 0.05$ (Table 2.9). Residual deviance for the final model for the complete dataset was 1140.1 (*df* = 1257, AIC score = 1156.1). Backwards stepwise regression analysis was used to assess the final model variables. Each term in the final model was tested with an analysis of variance (ANOVA) using a chi-square statistic (Table 2.10). The agriculture term (agr) achieves a reduction in deviance of 57.44 at a cost of 1 degree of freedom; the associated probability of that reduction by chance is ($p<0.000$). The final model (Response (*wetland presence*) ~ agr + trb + tri + riv + riv^2 + rds + agr:rds) indicates that the probability of wetland presence decreases with an increasing distance from tributaries, decreases rapidly with increasing terrain ruggedness, increases with increasing distance from river, decreases moderately with an increasing rate of decay away from surfaced roads, and decreases where land use is equal to agriculture (1) (Fig. 2.11). The interaction term including agriculture (agr) and roads (rds) in the model suggests that one factor is averaged over the effect of the other. Although the constituent term (rds) weakly suggests that the probability of wetland presence is greater near roads, the interaction term indicates that this probability may only be greater where land use is equal to agriculture. For non-agriculture, the probability of wetland presence increases with distance away from roads. Results from the full landscape model suggest that it is unlikely that such a difference in the distribution of actual wetlands versus random wetlands could have occurred by chance.

I calculated the odds ratio for wetland presence (95% CI) for each variable included in the final model (Table 2.11). The odds of a wetland being present increases by 0.30 times on non-agricultural land (95% CI. 0.22,0.41). For every 1 m distance increase away from tributaries (trb), the odds of a wetland being present decrease by 0.65 times (95% CI, 0.54, 0.79). For every 1 unit increase in terrain ruggedness (tri), the odds of a wetland being present decrease by 0.62 times (95% CI, 0.55,0.77). For every 1-m distance increase away from rivers (riv), the odds of a wetland being present increase by 1.39 times (95% CI, 1.17,1.65). For every 1 unit increase in transformed distance (negative exponential decay) from surfaced roads (rds), the odds of a wetland being present increase by 0.87 times (95% CI, 0.70,1.08). Wetlands were negatively related to agriculture land cover through the interaction between agriculture land cover and surfaced roads.

2.4.2.7 Model diagnostics for wetland occurrence

Sensitivity and specificity plots for the final model with a probability of wetland presence greater than 0.210, the proportion of correctly identified wetlands and incorrectly identified wetlands was 0.69 (Fig. 2.12a). The area under the ROC curve for the final model was 0.743, (Fig. 2.12b) indicating that the model had an appropriate ability to predict wetland presence.

2.4.3 Agricultural land versus non-agricultural land

To identify variables having effects on different land uses, I separated the data into agriculture (A1) and non-agriculture (A0), and modeled the new data separately.

2.4.3.1 Agricultural land use models (A1)

Residual deviance values, obtained from univariate analysis, ranged from 594.83 (trb) to 610.44 for the intercept term (Table 2.12). Pearson's correlation indicated a correlation between elevation and distance to river (r=0.813), distance to gravel roads and distance to structures (r=0.627), and terrain ruggedness and slope (r=0.951), thus three candidate covariates (elv, str, and slp) were excluded from the A1 model. Transformations (logarithmic, square-root and power) of the covariates showed some improvement (e.g. square-root transformation of TRB), but none were noticeable enough to justify the use of transformations. From the univariate analysis, distance to tributaries (trb) was the most discriminating variable with the lowest residual deviance and was therefore selected as the first step of the stepwise procedure. The modeling procedure resulted in two final models shown in Table 2.13a and 2.13b. The first model (A1.1*)* included two terms (trb and rds). The second model (A1.2) included two terms (trb and riv). Residual deviance and AIC values for both candidate models (A1.1 and A1.2) are given in Table 2.14. Analysis of variance values of chi-square for model terms are shown in Table 2.15a for model A1.1 and in Table 2.15b for model A1.2.

2.4.3.2 Non-agricultural land use models (A0)

Residual deviance values, obtained from univariate analysis, ranged from 598.08 (slp) to 630.7 for the intercept term (Table 2.16). Pearson's correlation indicated a correlation between elevation and distance to river (r=0.842), distance to gravel roads and distance to structures (r=0.666), and terrain ruggedness and slope (r=0.983), thus three candidate covariates (elv, str, slp) were excluded from the A0 model. Transformations (logarithmic, square-root and power) of the covariates showed some improvement (e.g. square-root transformation of TRB), but none were noticeable enough to justify the use of transformations. From the univariate analysis, terrain ruggedness (tri) was the most discriminating variable with the lowest residual deviance and was therefore selected as the first step of the stepwise procedure. The modeling procedure resulted in five final models shown in Table 2.17 (a-e). All five models (A0.1, A0.2, A0.3, A0.4, and A0.5) included the term for terrain ruggedness (trb). Additional terms for each model were: cti (A0.1), riv (A0.2), rdg (A0.3), trb (A0.4), and rds (A0.5). Analysis of variance values of chi-square for model terms are shown in Table 2.18.

2.4.3.3 Combined land use models

Model variables from agriculture and non-agriculture that overlapped were combined to create one model to explain and compare both landscapes. Three variables were included in the overlapping and final model for the agricultural (A1) and non-agricultural land class (A0): (1) distance to tributaries (trb), (2) terrain ruggedness (tri), and (3) distance to river (riv). Model output for agricultural land (A1) is shown in Table 2.19 and in Table 2.20 for non-agriculture land (AO). The analysis of variance values of chi-square for each term in model A1 are shown in Table 2.21 and in Table 2.22 for model A0.

2.5 Discussion

2.5.1 Wetland distribution

The majority (160/265) of wetlands occurred on lands classified as non-agricultural, yet non-agricultural represented only 34.5% of the available land cover. The representational imbalance of wetland presence may be a result of the historical practice of draining wetlands from land deemed suitable for agriculture. Environment Canada (1986) has estimated that at least 75 percent of previously existing wetlands on the Canadian prairies and parkland have been lost through agricultural land conversion. Wetlands occurring on non-agricultural land may represent true pre-settlement conditions but may also be over-represented by wetlands displaced via drainage from agricultural areas. The abundance of wetlands on non-agricultural land may have been underestimated due to the difficulty of detecting wetlands under forest cover on aerial photographs. Lastly, wetland persistence fluctuates with local climate and a general trend of warming and drying across the Canadian prairies since the early 1960s suggests that the ability of the landscape to sustain wetlands is steadily decreasing.

2.5.2 Factors affecting wetland presence

Model explanation of wetland occurrence

Wetland presence was explained, in part, by surfaced roads (rds) but not private, lease or non-surfaced roads (rdp, rdl, rdg), and this may be due to the distribution of these road types on the study landscape as they are not equally represented or distributed. Most surfaced roads are located systematically by the Alberta Township System (ATS: used to divide Crown land into private parcels) every one mile apart in the north/south direction and every two miles apart in the east/west direction in a grid-like pattern (Government of Alberta). The density of surfaced roads in the core area of the study landscape overlap with agricultural land (from which wetlands have been drained) and this likely contributes to the high probability of finding a wetland within a short distance of a road, or a road near any single wetland. The probability of wetland presence was greater on non-agricultural land which is most often located away from roads. Accordingly, wetlands on this land cover are also more likely to be found away from roads. The remaining road types (private, lease, and non-surfaced) are sparse, aggregated and likely constructed where physical land features (e.g. higher and drier areas) were most conducive to road construction and not restricted to the ATS grid.

With increasing distance from tributaries (trb), the probability of wetland presence decreased. Wetlands are likely to occur near tributaries because both are determined by similar geomorphic and hydrologic conditions and because tributaries exist where slope discontinues (Bedford, 1996). Snow pack is likely be greater near tributaries than on agricultural fields where winter vegetation is sparse and snow is exposed to wind and greater rates of evapotranspiration; spring melt would thus contribute to wetlands in lower elevation areas and near tributaries. Wetlands near tributaries may also be remnants of the pre-settlement landscape that were spared from agricultural land conversion due to their position on land not suitable (too steep) for agriculture. Wetlands adjacent to tributaries may play an important role in the chemical, physical and biological integrity of hydrologic systems downstream of the tributary and should therefore be retained, protected, or restored where possible. To exemplify the importance of wetlands near tributaries, Gilmor (2004) suggests that the restoration of wetlands near tributaries feeding the headwaters of the Red River in Manitoba, Canada could improve river water quality by attenuating spring run-off and reducing agricultural (pollutant) inputs into the river. The same logic could be used for Genesee where tributaries feed into the North Saskatchewan River.

With increased terrain ruggedness (tri), the probability of wetland presence decreased rapidly. As a variable created from an algorithm using slope, terrain ruggedness was highly correlated with slope and this is emphasized by the nearly indistinguishable feature classes (tri and slp) in GIS. Accordingly, this variable suggests that wetland presence decreases with increasing slope and for Genesee, steeper slopes are equated with river valleys and ravines where drainage would override input and surface water retention. Slope and ruggedness should be distinguishable components of the terrain and Sappington et al. (2007) recommend an alternative method to create a vector ruggedness measure (VRM) of terrain that is less correlated with slope. The separation of these two components for analysis would allow for a better understanding of how slope and ruggedness, as independent and as combined characteristics of the terrain, affect wetland presence. Spatial uncertainty is also assumed when using a 30m DEM to derive terrain characteristics, particularly in this research where landscape entities being analyzed are less than 30 m^2.

Wetland presence increased with increasing distance from rivers (riv), but this relationship may be explained more by site-specific variables such as soil (texture, parent material, moisture) and groundwater level. At the landscape scale, this observation may be explained best by three variables: slope, elevation and land cover. First, the highest elevation (meters) on the landscape is also the furthest from rivers and accordingly, these two variables (riv and elv) were highly correlated. More wetlands at this higher elevation could be a result of a plateau where natural wetland drainage is minimal. Wetlands nearer to the rivers may be exposed to a greater slope within the drainage basin and may therefore be more prone to discharge into the river. Campling et al. (2002) modeled the probability of soil drainage class and found significant relationships between soil drainage and three spatial determinants: elevation, slope, and distance-to-river; though they regarded distance-to-river as “a surrogate terrain variable as it integrates the terrain and landform information inherent to the measure of river channel proximity”. The terrain variables that they refer to as determinants of soil drainage class are slope and elevation. Therefore, distance to river alone is difficult to isolate as a single determinant of wetland presence. Lastly, it is possible that wetland presence near the river was underestimated due to greater forest cover (cartographic error).

Remaining variables that were not significant in explaining wetland presence were sinks (snk) and soil wetness (cti). The extremely low presence of sinks defined by the DEM on the study landscape reduces the probability of any point (random or actual wetland) co-occurring with a sink. Sink features may have a greater effect in studies examining larger areas. Wetland presence increased with increased soil wetness (cti) and this was significant ($p<0.0001$) in the univariate model suggesting that this variable would predict wetland presence. However, with the forward stepwise selection of variables for the multivariate model, cti did not contribute to model fit as expected and was not included in the final model. Further research would be necessary to examine the relationship between cti and the variables included in the final model to understand the isolated effects of one variable versus the compound effects of many variables.

2.5.3 Agricultural land versus non-agricultural land

In the final model, the variable land use (agriculture or non-agriculture) was statistically significant in explaining wetland geographic distribution, while interaction terms including agriculture land use were not significant. Without a model, a simple observation of the aerial photographs would also suggest, to some measure, that wetlands presence is greater on non-agricultural land. To understand the factors regulating the presence of wetlands on agricultural land versus non-agricultural land, it was necessary to explore effects at both levels of land use separately. Analysis at this level was justified to examine the effects of the variables by land use and obtain a more detailed understanding of the factors influencing wetland presence.

The magnitude of influence of 'distance to rivers' (riv) was relatively greater on agricultural land ($p<0.01$) than non-agricultural land ($p<0.05$). As illustrated in the discussion of the full landscape model, the furthest distance from the river is spatially coincident with a plateau at the highest elevation on the landscape. The fact that this area is mainly agricultural land is probably not by chance but rather by bias selection of land, flat land, suitable for agriculture. The presence of wetlands on agricultural land may mean that site-specific factors such as soils and hydrology (e.g. position of the water table) are conducive to wetland persistence and reestablishment even where wetland drainage has been attempted. If slope was greater on this part of the landscape, I would expect it to influence drainage more so on agricultural land than on non-agricultural land because farmed land is generally uniform, more exposed to sun and wind and less protected from erosion by woody plants.

Terrain ruggedness (tri) was negatively related to wetland presence for agriculture (A1) and non-agriculture (A0), but had a relatively greater influence on wetland presence for the non-agricultural land ($p<0.001$). The homogenous nature of the agricultural land reduces the available range of ruggedness index values relative to the available range of values for the non-agricultural land and helps to explain why this particular effect has a limited effect on agricultural land. The greater percentage and availability of rugged areas on non-agricultural land, considered too rugged for wetland presence, may explain the greater variation between wetland presence and absence for this land cover. As discussed with the full landscape model, terrain ruggedness is correlated with slope and both variables increase in value where steeper ravines drain into tributaries or river channels which mainly exist on non-agricultural land.

Distance to tributaries was negatively related to wetland presence on agricultural land ($p<0.001$) more so than on non-agricultural land ($p<0.01$). The co-occurrence of these hydrologic features may be explained by geomorphology. If a tributary exists on, and is surrounded by crop land, it is likely because the topographic and geomorphic conditions (e.g. slope, soil wetness, groundwater) determining the tributary are persistent enough to override a land owner's attempts to convert it to farm land. Site conditions near tributaries may have allowed for wetlands that were too large to fill or too deep to drain (Biebighauser 2007), and wetlands near tributaries (in the riparian zone) may persist because it was too difficult or expensive for the landowner to justify drainage. Lastly, as mentioned with the full model, the presence of wetlands near tributaries may be attributed to greater snowpack relative to the open and surrounding agricultural land where snow depth is limited by greater exposure to wind and sunlight.

Prediction map

The prediction-based continuous maps of wetland presence are presented in Figure 2.13. In this map, the probability that a wetland occurs in a given cell is represented by values between 0 and 1, where 0 (colour = red) represents a low probability and 1(colour = green) represents a high probability of wetland presence. The wetland probability map shown represents model predictions where anthropogenic influences (roads and agriculture) have been removed and indicates where wetlands may have persisted based on the three remaining model covariates: distance to rivers (riv), terrain ruggedness (tri), and distance to tributaries (trb). The use of the cut-off value (0.21- determined form the sensitivity/specificity curves) to reclassify the prediction map resulted in the majority of the landscape interior being reclassified as '1' – likely to be occupied by a wetland according to the model. This simplified map did not provide insight to specific sites, thus the final prediction map includes all possible cut points for interpretation. Without roads and agricultural land use, the three remaining variables in the model, related to terrain and natural hydrological features, may be too homogenous throughout the interior of the landscape to distinguish specific wetland sites. Site specific soil data would likely help to isolate potential wetland sites, given the remaining variables in the model (riv, tri, and trb).

2.6 Conclusions and Recommendations

A pre-disturbed landscape state does not always equate to desirable state for restoration. Results from my model suggest that wetlands on the pre-mined Genesee landscape were determined mainly by agricultural land use, associated roads and terrain characteristics attributed to the dominant land use. Characterization of the pre-mined state of wetlands is intended to provide a baseline against which future landscape plans can be evaluated. For Genesee, the potential exists for the landscape to include productive agricultural systems and an ecologically interactive natural wetland ecosystem.

Results from my study indicate that the probability of wetland presence is greater on non-agricultural land which represents a small fraction of the available landscape. These results may be due to a selection bias because the random selection of sites for non-wetland points did not represent every available site within the extents of the study area. Based on the results, I conclude that 1) the land classified as non-agriculture on the pre-mined landscape is representative of what the majority of the landscape may have looked like pre-settlement 2) the potential for wetlands to occur on the pre-mined landscape has been underestimated, 3) without pre-settlement data, particularly large-scale aerial photographs, models are limited by available data, 4) future wetland site assessments should include site-specific soils data, 5) wetland presence on the pre-mined landscape is determined mainly by agricultural land use and related terrain characteristics, and 6) wetland restoration efforts at Genesee should focus on the potential for land use to affect wetland persistence for each site selected by its site-specific characteristics in conjunction with landscape scale characteristics. My research model is supported by Zedler (2003) who suggests that with respect to wetlands, "scientists can help plan restoration by adapting existing landscape design models to agricultural landscapes, proposing alternative strategies, and evaluating their effectiveness".

Results from this study can be used to aid restoration efforts by identifying which patterns from the historical landscape should be replicated and which could be improved on. The methods used in this study are not intended to replace the necessary on-the-ground research and analysis, but rather to serve as an additional tool that can be used to reduce field work, delete measurements unlikely to yield useful information, and spotlight knowledge needs (such as soil classifications) to facilitate successful wetland restoration. It is important to recognize that the wetlands in this study were represented by ‘points’ rather than polygons which suggests a misrepresentation of the land feature as a spatial unit. This effect may be negligible for Genesee where the average wetland is very small but may affect studies involving larger wetlands. Although the approach used for this study does not guarantee wetland success, it may be the only feasible option, at this time, for recreating and quantifying historical landscapes and establishing a historical range of variability for a particular landscape.

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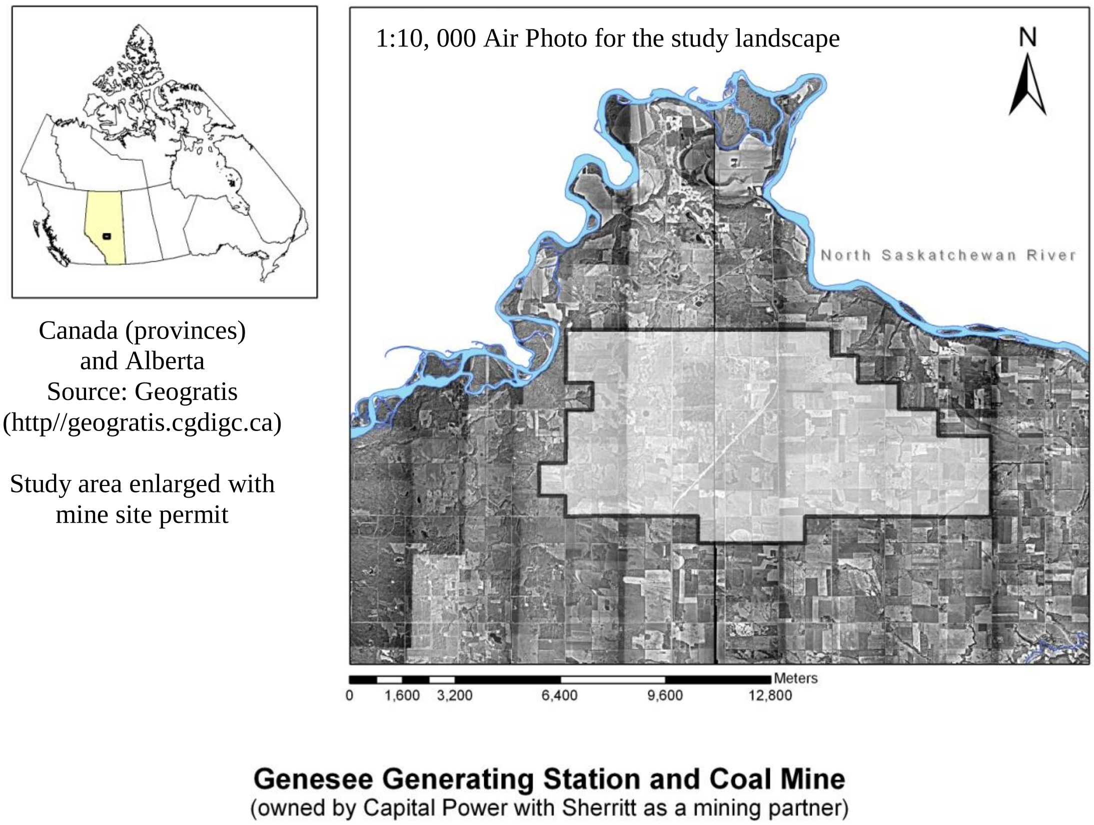


Figure 2. 1 Image showing extents of the study location. The Genesee Generating Station and coal mine site is located approximately 70km south west of Edmonton, Alberta Canada. The present-day mine permitted area is shown here (shaded box outlined with black) on top of the historical (1982) study landscape.

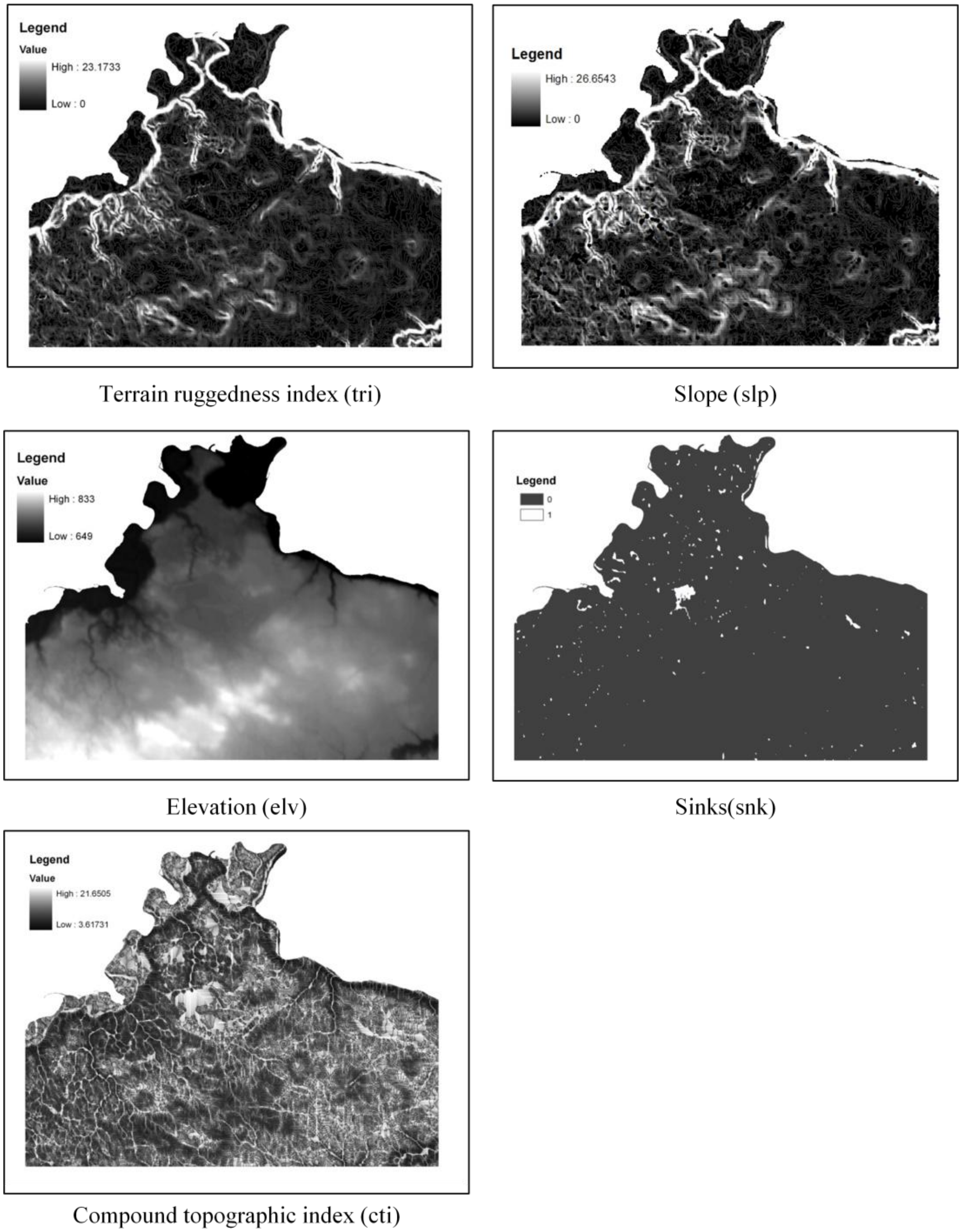


Figure 2. 2 Five feature classes (tri, slp, elv, snk, cti) derived from the digital elevation model (DEM) to represent candidate covariates explaining wetland presence.

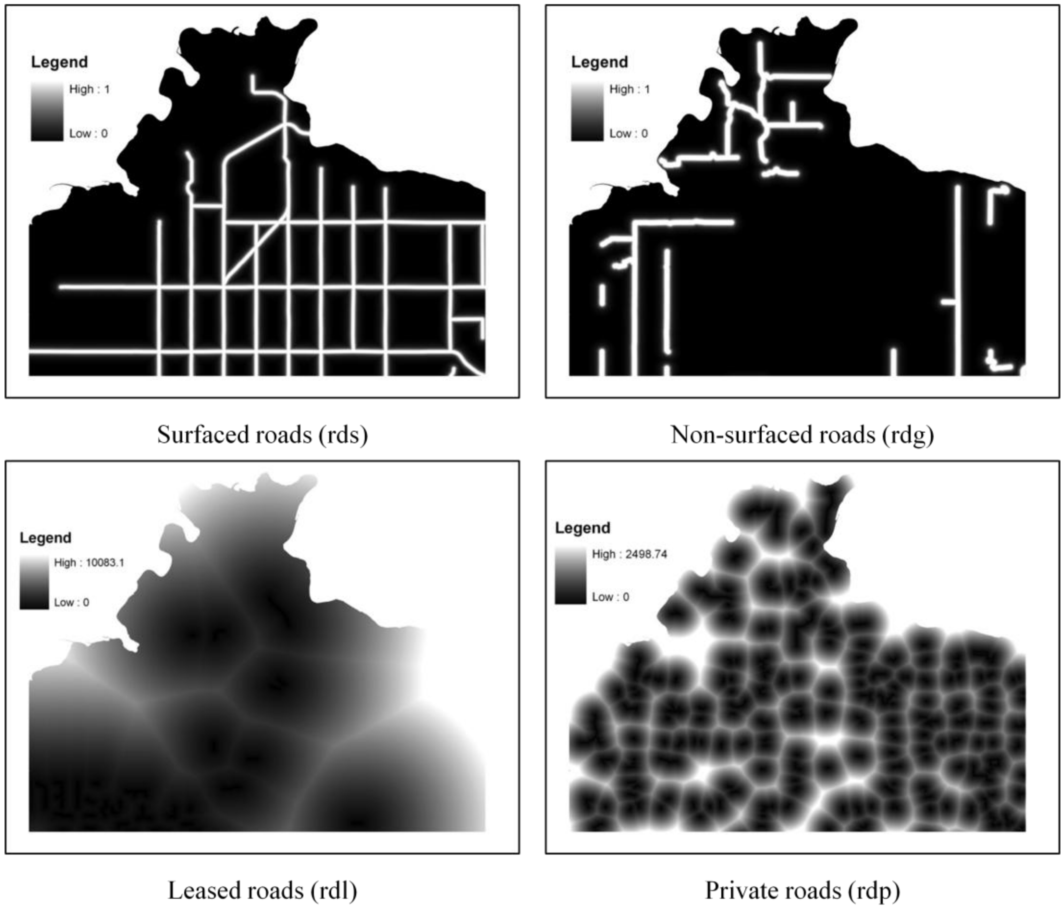


Figure 2. 3 Eight feature classes (rds, rdg, rdl, rdp, agr, trb, str, riv) derived from aerial photograph interpretation to represent candidate covariates explaining wetland presence.

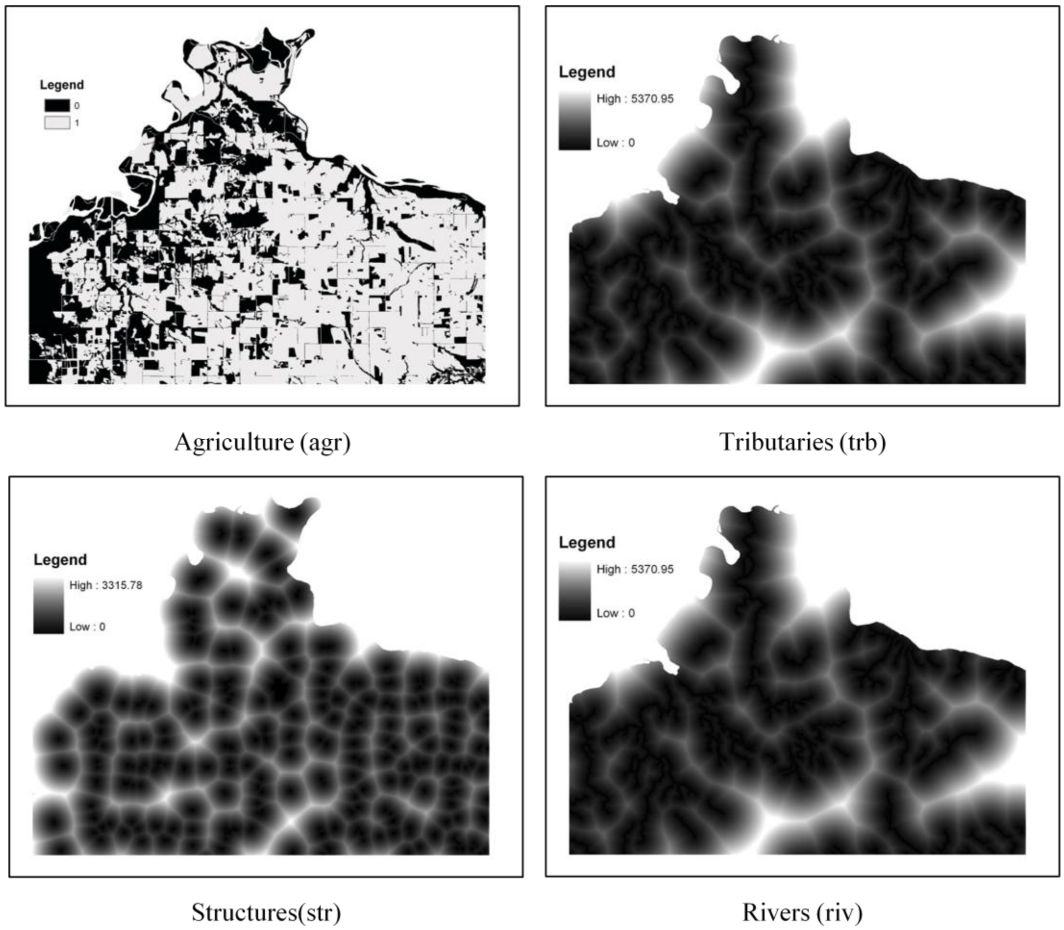


Figure 2.3 Continued. Eight feature classes (rds, rdg, rdl, rdp, agr, trb, str, riv) derived from aerial photograph interpretation to represent candidate covariates explaining wetland presence.

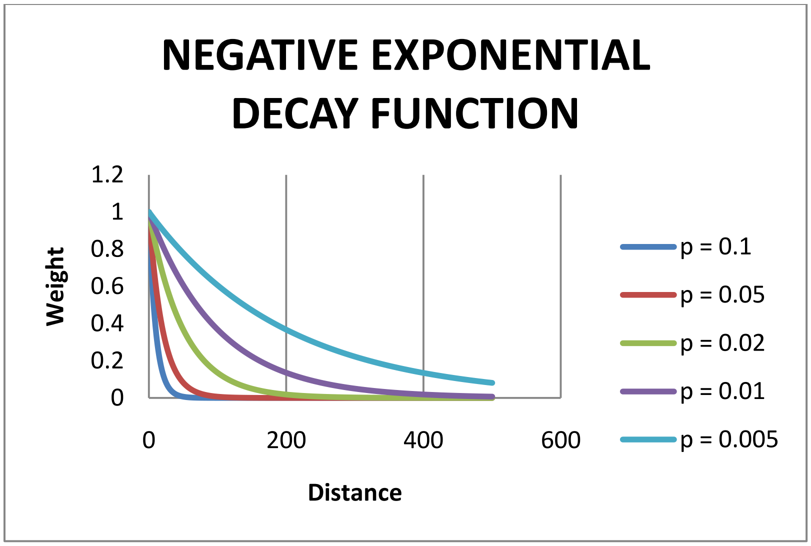
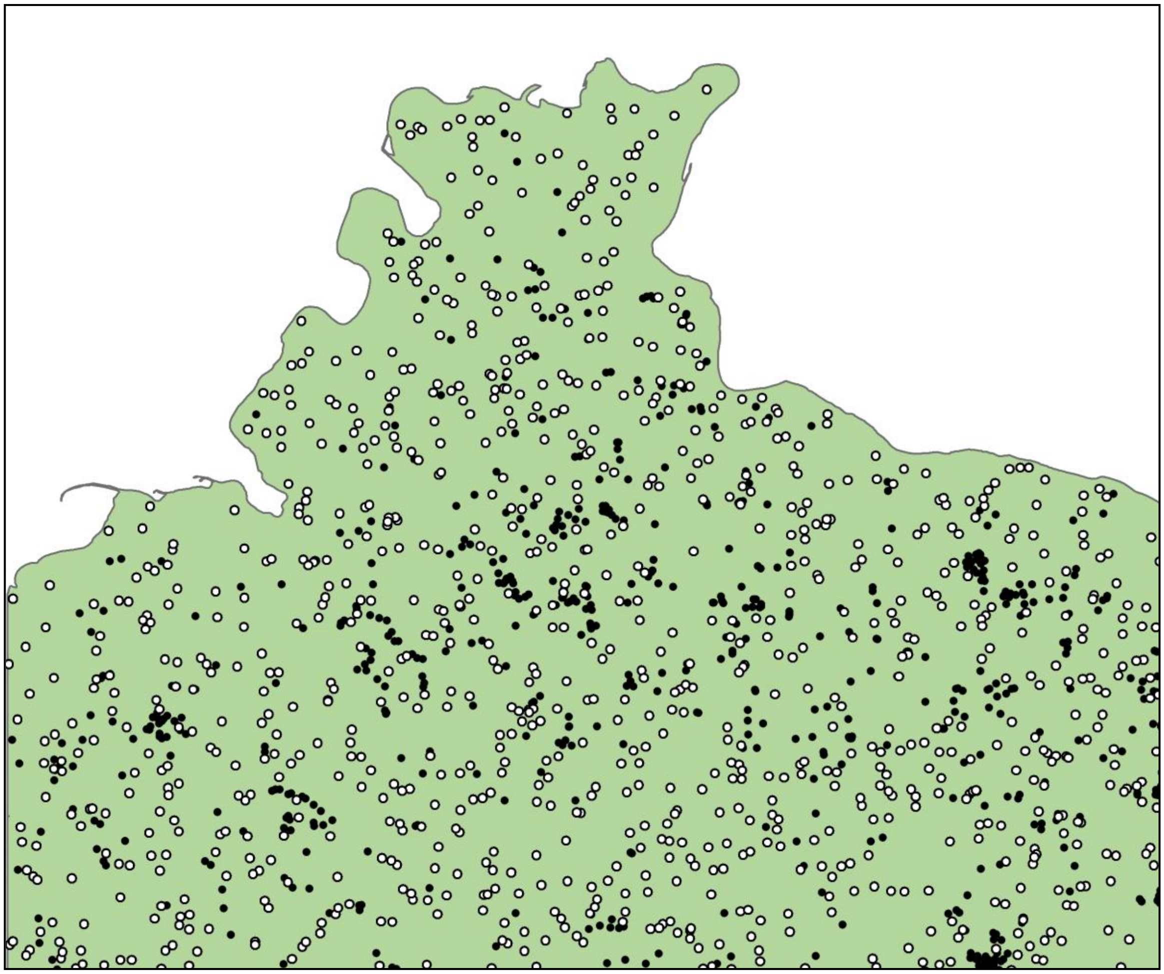


Figure 2. 4 Graph showing negative exponential decay function for decaying distance from road feature. At a decay function (p=0.1), most road effects (~95%) diminish at ~300 m.



- ● Wetland points (265)
- ○ Random points (1000)

Figure 2. 5 Image showing study landscape with 1000 random points and 265 true wetland points

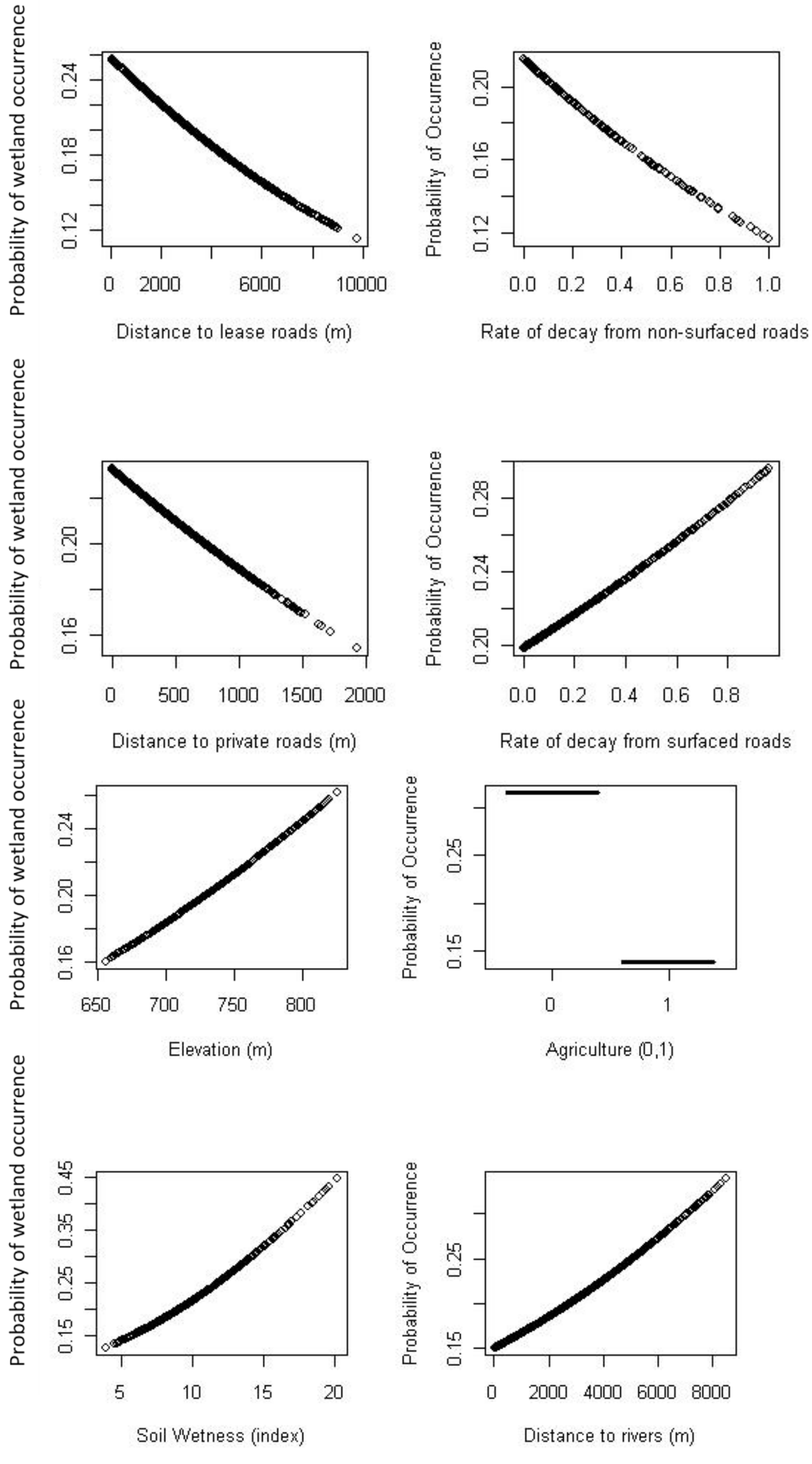


Figure 2. 6 Plot of estimated probability of [wetland] occurrence from univariate analysis of thirteen candidate covariates. Continued on following page.

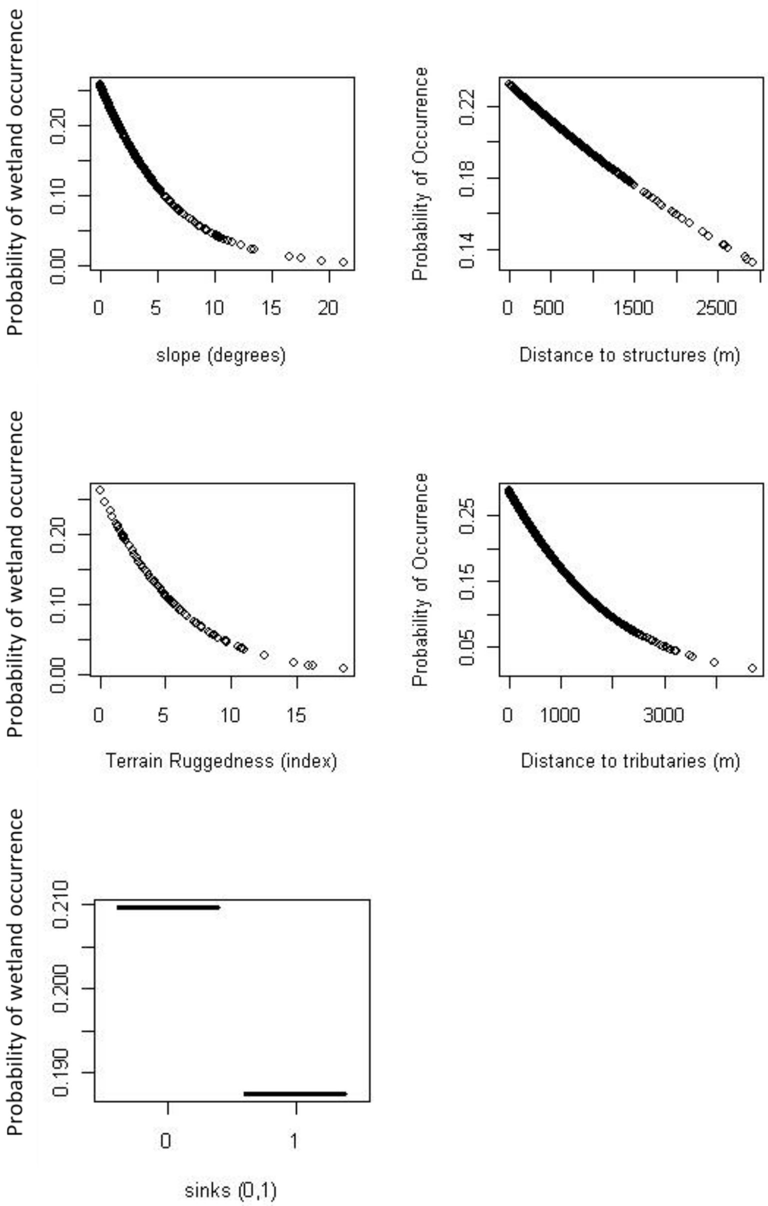


Figure 2.6 continued. Plot of estimated probability of [wetland] occurrence (fitted values) from univariate analysis of thirteen candidate covariates.

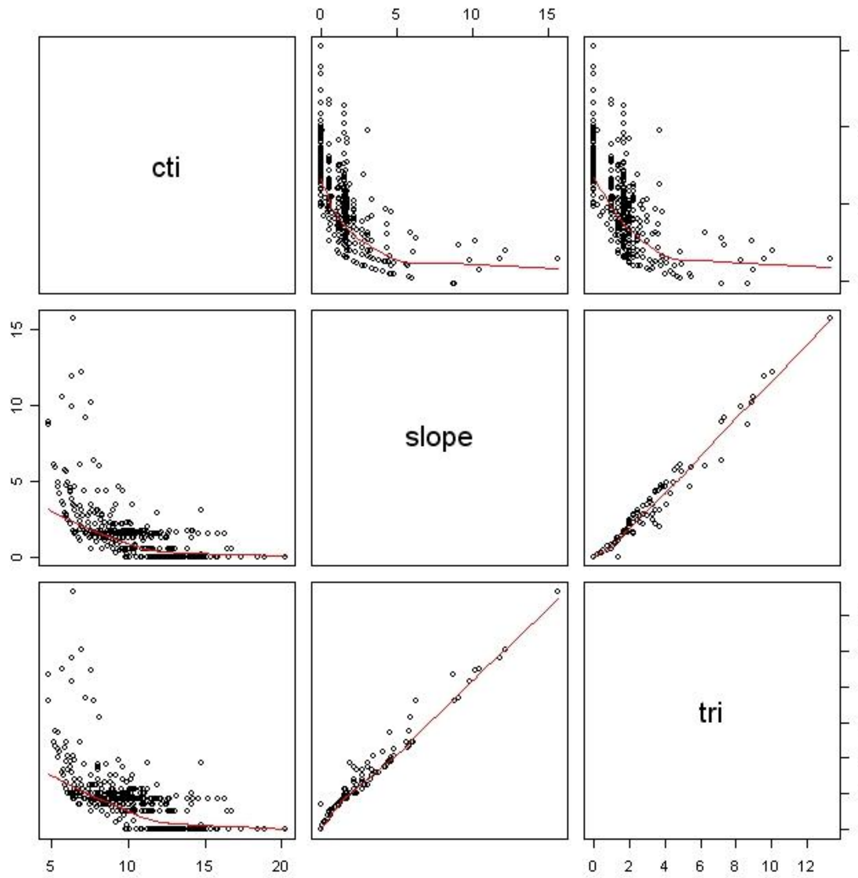


Figure 2. 7 Scatter plot matrix showing correlated covariates (CTI, SLOPE, TRI). SLOPE and TRI are highly correlated (r=0.974); slope and CTI are moderately and negatively correlated (r=-0.502); CTI and TRI are moderately correlated (r=-0.536).

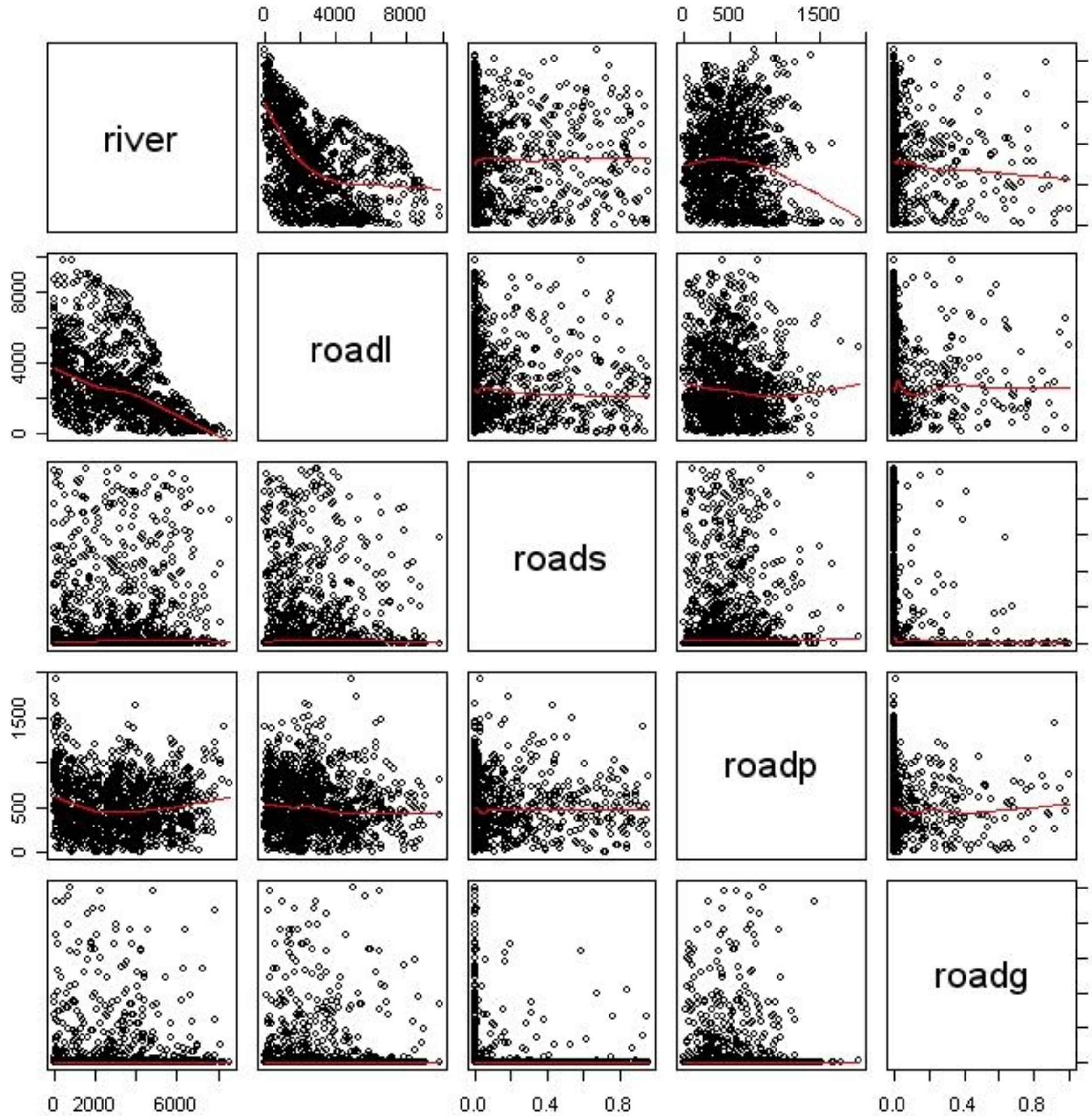


Figure 2. 8 Scatter plot of predictor variables demonstrating weak correlations (r < +/- 0.475): distance to river(riv), distance to lease road (rdl), surfaced road (rds), private road (rdp), non-surfaced road (rdg).

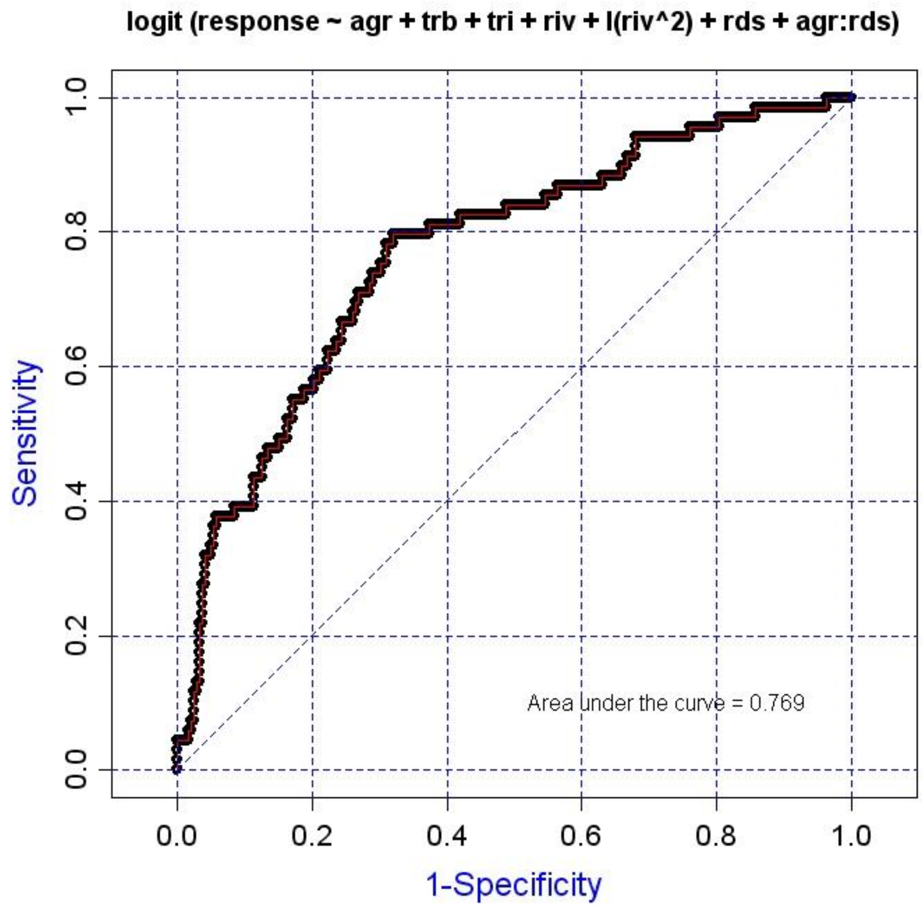


Figure 2. 9 Plot of the ROC (sensitivity versus 1-specificity) for all possible cut points (0.2 increments) in the testing regression model explaining the influence of model covariates on wetland presence. The area under the curve is approximately 0.769.

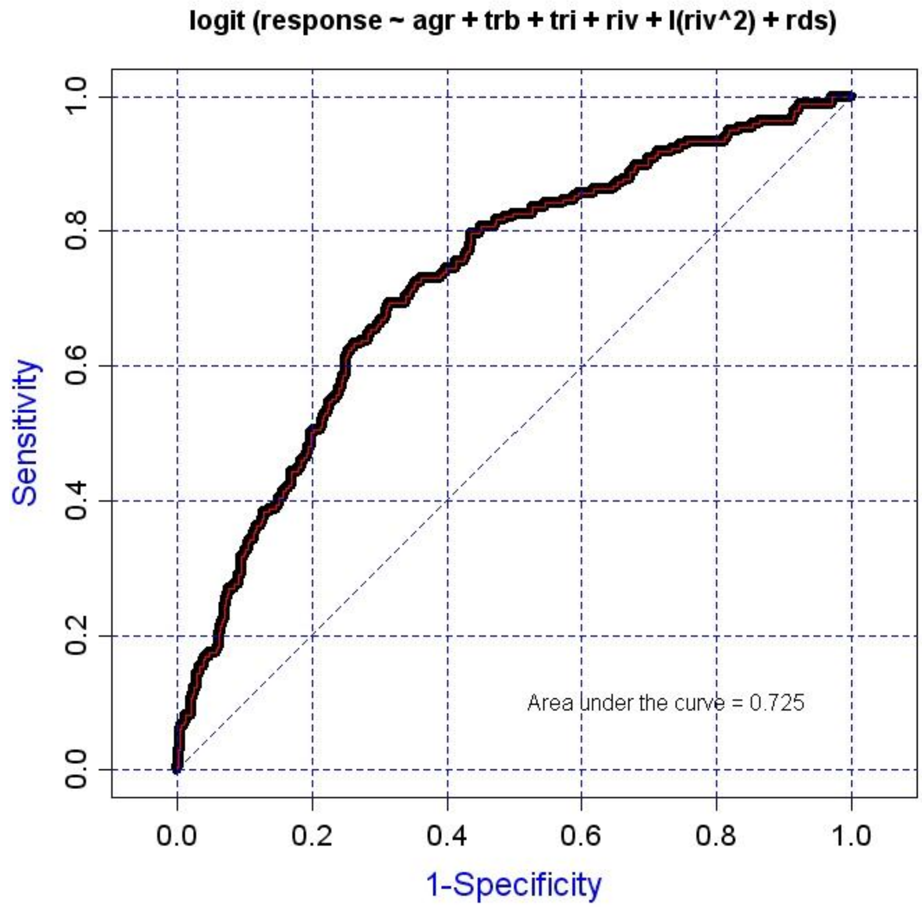


Figure 2. 10 Plot of the ROC (sensitivity versus 1-specificity) for all possible cutpoints (0.2 increments) in the training regression model explaining the influence of model covariates on wetland presence. The area under the curve is approximately 0.725.

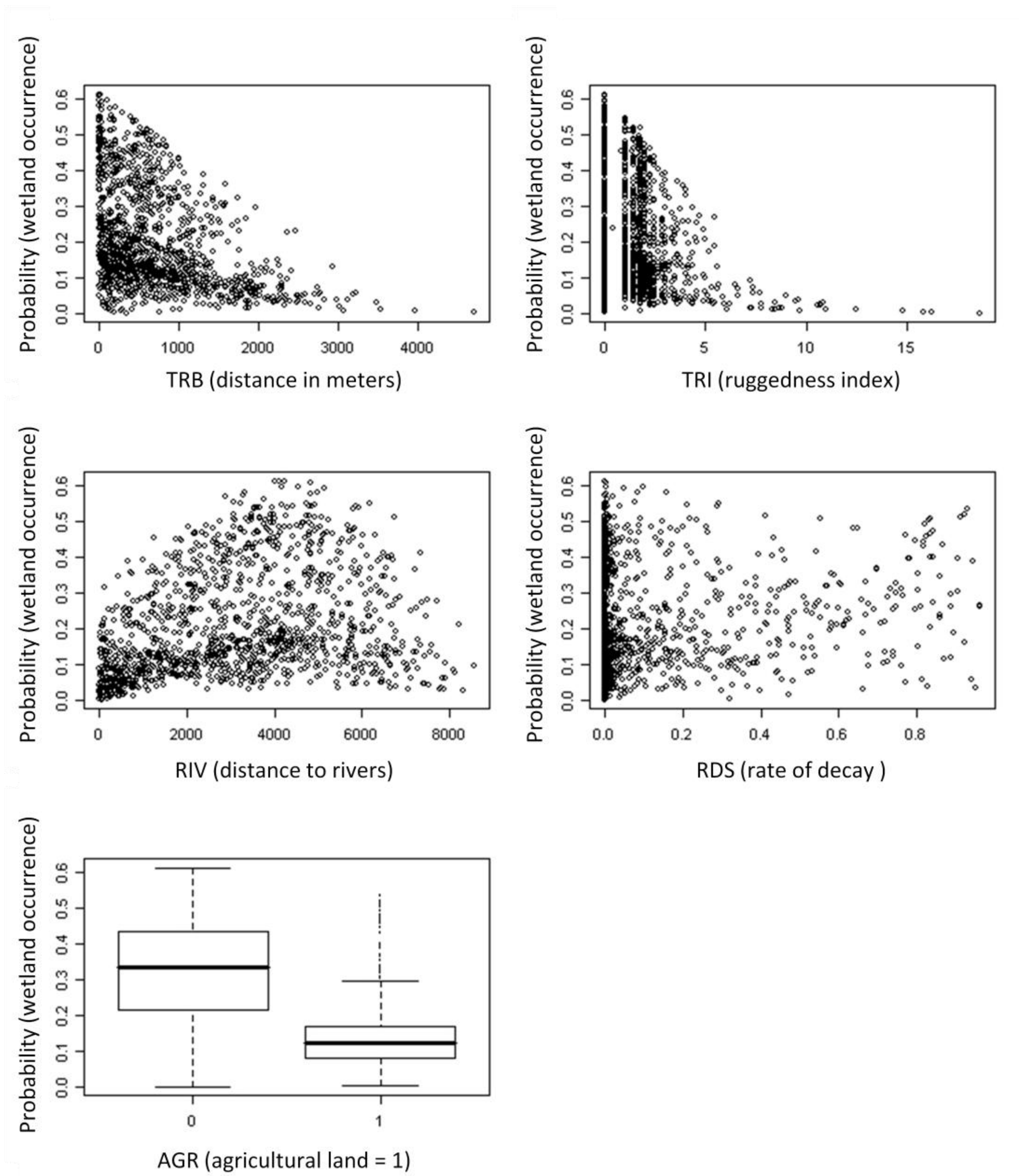


Figure 2. 11 Plot of five variables included in the final fitted model showing the probability of wetland occurrence by distance to tributaries (trb), terrain ruggedness (tri),distance to rivers (riv), distance to surfaced roads (rds), and land use (agr), where agricultural land = 1.

a)

b)

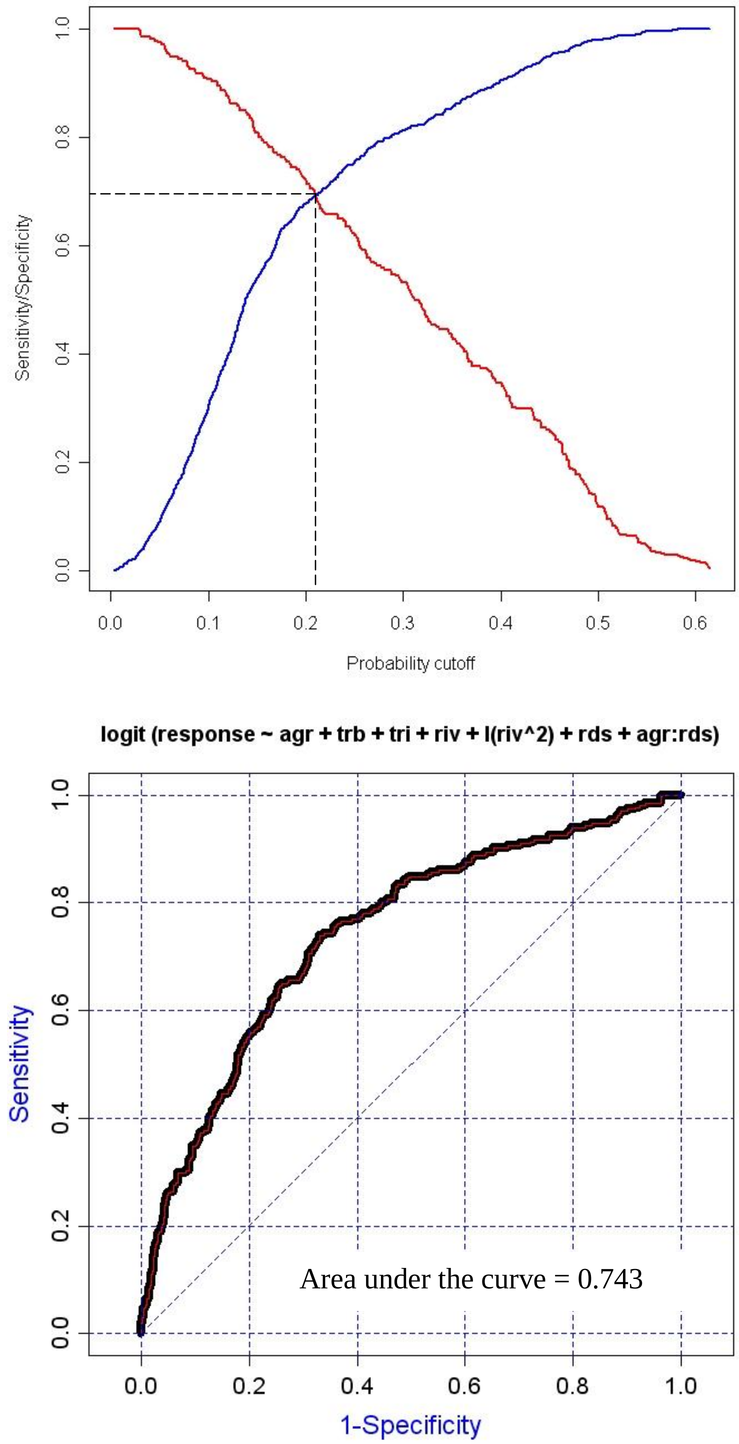


Figure 2. 12 Plot of a) sensitivity and specificity versus cutpoints (0.1 increments). Using a cutoff of 0.21, the sensitivity/ specificity is 0.69 and b) the ROC (sensitivity versus 1-specificity) for all possible cutpoints (0.2 increments) in the final logistic regression model explaining the influence of model covariates on wetland presence. The area under the curve is approximately 0.743.

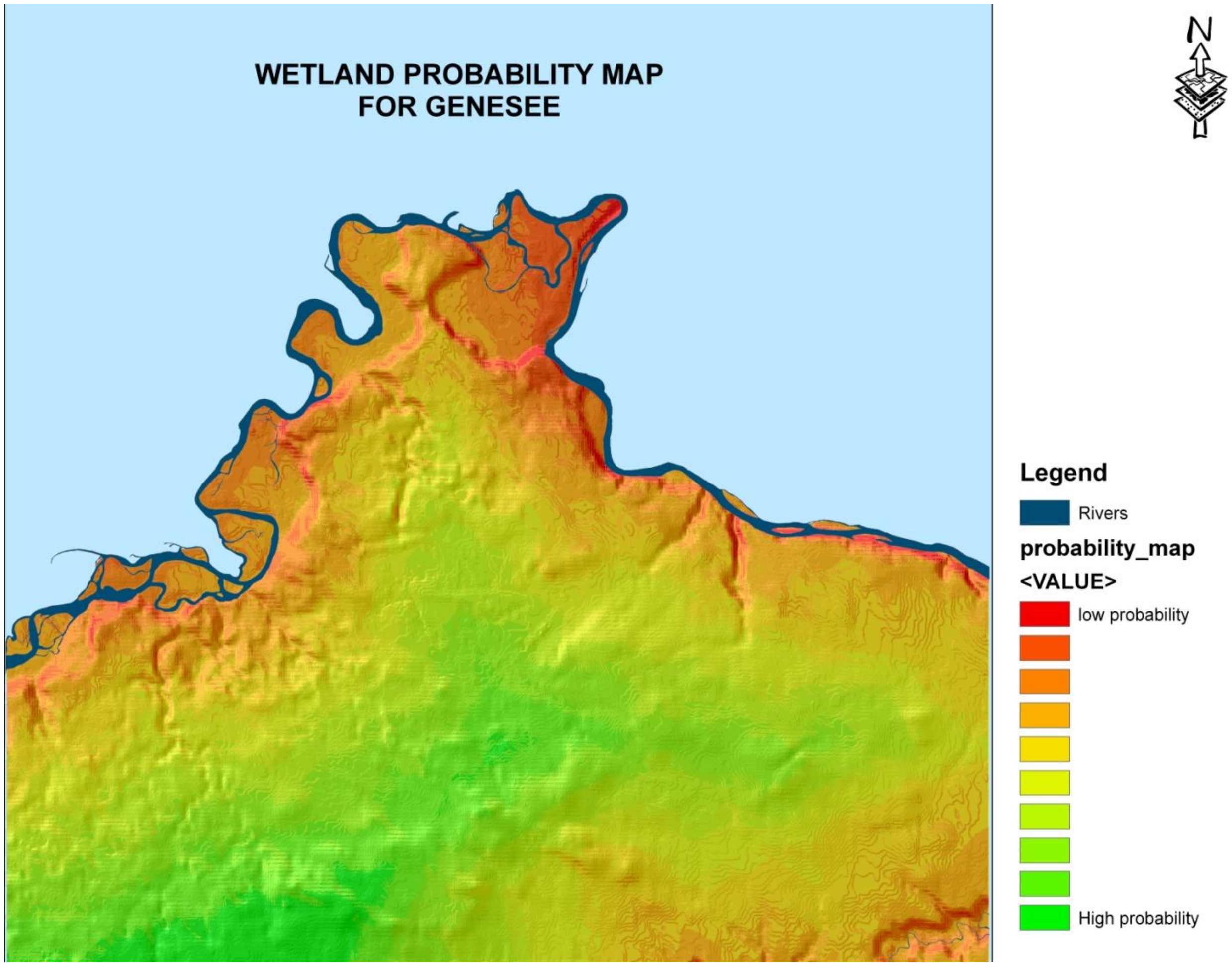


Figure 2. 13 Map showing final model probabilities of wetland occurrence on the pre-mined landscape where two variables (roads and agricultural land) have been removed to simulate a landscape before settlement. Each cell is represented by values between 0 and 1, where 0 (colour = red) represents a low probability and 1(colour = green) represents a high probability.

Table 2. 1 Example from literature of factors affecting wetland services.

Factor affecting wetland services	Description	Authors
Degree of slope	Slope determines the degree of interaction between surface and ground water which affects surface water depth.	Kazezyilmaz-Alhan and Medina 2008
Topography	Topography within a catchment directly influences dissolved organic carbon flux which indirectly affects stream water chemistry	Andersson and Nyberg 2008
Proximity to streams	Wetlands near streams can serve as pollution control systems via the process of denitrification which decreases the amounts of nitrogen entering stream channels	Kaushal et al. 2008
Proximity to seed sources	The proximity of wetlands to seed-sources affects wetland plant composition by altering the abundance and distribution of plant propagules and the number of propagule-dispersal routes, and wetlands near forests are less likely to contain invasive species because the forest acts as a buffer.	Houlahan et al. 2006
Proximity to roads	wetlands near roads may contain as much as 4000mg/l of road salt (NaCl, CaCl and MgCl) which has been shown to alter wetland plant communities and negatively impact resident amphibians	Environment Canada 2001 Richburg et al. 2001 Karraker et al. 2008 Sanzo and Hecnar 2006

Table 2. 2 Descriptions of the land use and land cover (LULC) categories derived from the historical aerial photographs using GIS.

Land cover feature classes	***Description***
Agriculture	agronomic-related features such as farmsteads, crop land and pastures. Cell value = 1
Non-agriculture	All land not classified as agronomic. Includes deciduous, coniferous, and mixed forests, as well as grassland and shrub/willow covered areas. Cell value = 0
Structural	Buildings – mainly dwellings and agriculture-related structures. Converted to a raster; each cell represents a euclidean distance in meters from the nearest structure.
Roads:	
Surfaced roads	Township roads - graded and surfaced (e.g. gravel or pavement). Converted to a raster; each cell represents an exponential decay value from surfaced roads (1 to 0); value of 1 at the road.
Non-surfaced roads	roads – non-surfaced. Converted to a raster; each cell represents an exponential decay value from non-surfaced roads (1 to 0); value of 1 at the road.
lease	Private access lease roads (associated with oil/gas companies). Converted to a raster; each cell represents a euclidean distance in meters from the nearest lease road.
private	Private entries / access farmsteads. Converted to a raster; each cell represents a euclidean distance in meters from the nearest private road.
Rivers and streams	rivers (assigned to the North Saskatchewan River) and streams (Strawberry Creek). Converted to a raster; each cell represents a euclidean distance in meters from the nearest stream or river.
Tributaries	assigned to all channels feeding into either the North Saskatchewan River or Strawberry Creek. Converted to a raster; each cell represents a euclidean distance in meters from the nearest tributary.

Table 2. 3 Table showing feature class codes (acronyms), units of measurement (m= meters, 0/1=binary, NED=negative exponential decay parameter, %= percent), description (C= continuous data, D=discrete data), and raster cell values (minimum and maximum).

Feature Class	*Code*	*Unit*	*C/D*	*format*	*Min*	*Max*
Agriculture	agr	0/1	D	Binary	0	1
Structures	str	m	C	stretch	0	3315.78
Roads:						
surfaced	rds	NED	C	stretch	0	1
non-surfaced	rdg	NED	C	stretch	0	1
lease	rdl	m	C	stretch	0	10083.09
private	rdp	m	C	stretch	0	2498.73
Rivers and streams	riv	m	C	stretch	0	8609.10
Tributaries	trb	m	C	Stretch	0	5370.95
Sink	snk	0/1	D	Binary	0	1
Slope	slp	%	C	stretch	0	26.65
Elevation	elv	m	C	stretch	649	833
Soil wetness	cti	index	C	stretch	3.62	21.65
Terrain ruggedness	tri	index	C	stretch	0	23.17

Table 2. 4 Land cover classes for Genesee where total land cover is classified as either agricultural land (65%) or non-agricultural land (35%).

BROADCLASS	**Area (ha)**	**percent cover**
Agricultural land total	18421	65%
Sand/Gravel	78.46	0.3%
Coniferous	320.50	1.1%
Deciduous	4699.01	16.5%
Grassland	430.15	1.5%
Mixed	2707.35	9.5%
Shrub/Willow	1654.42	5.8%
Windbreak	131.81	0.5%
Non-Agricultural land	10022	35%
TOTAL LANDSCAPE AREA	**28443**	**100.0%**

Table 2. 5 Data points equal to true wetland points (1) and non-wetland points (0) on agricultural land versus non-agricultural land.

	Land Use		
Data points	**Agriculture**	**Non-Agriculture**	**TOTAL**
True wetland points (1)	105 (40%)	160 (60%)	265
Random points (0)	655 (65.5%)	345(34.5%)	1000
TOTAL	**760**	**505**	**1265**

Table 2. 6 Univariate analysis using training data to rank candidate covariates for model approximating wetland presence on the pre-mined landscape. Shown are the parameter estimates with standard error and univariate Wald test statistics. Variables are sorted by residual deviance values (Rdev); smallest to largest.

	Estimate	**Std.Error**	**z_value**	**Pr(|z|)**	**Rdev**
agr	-0.92040	0.16170	-5.691	0.00000	**952.63**
trb	-0.44554	0.10054	-4.432	0.00001	**962.40**
slp	-0.46880	0.12780	-3.666	0.00025	**967.34**
tri	-0.40672	0.11216	-3.626	0.00029	**969.34**
rdg	0.27826	0.07587	3.668	0.00025	**972.52**
riv	0.24455	0.07882	3.103	0.00192	**976.07**
cti	0.23978	0.07724	3.105	0.00191	**976.17**
rds	-0.22901	0.09154	-2.502	0.01240	**978.77**
rdl	-0.19224	0.08438	-2.278	0.02270	**980.28**
elv	0.11833	0.08040	1.472	0.14100	**983.52**
str	-0.10856	0.08397	-1.293	0.19600	**983.97**
rdp	-0.07809	0.08072	-0.967	0.33300	**984.77**
snk	-0.31015	0.77867	-0.398	0.69000	**985.55**
(Intercept)	-1.30286	0.07915	-16.460	<2e-16	**985.71**

Table 2.7 Parameter estimates using the standardized values for variables with coefficients, standard error, Z statistic (standardized normal deviate), and p-value for the best approximating model for wetland presence (including 754 random points and 196 wetland points) for the training dataset (75% of the full dataset) . Residual deviance for the final model for the training dataset was 856.726 on 942 degrees of freedom (AIC score = 872.72).

	Estimate	Std. Error	z value	Pr(>|z|)
(Intercept)	-0.55276	0.14723	-3.754	0.000174
agr1	-1.15738	0.17680	-6.546	5.9e-11
trb	-0.38073	0.10673	-3.567	0.000361
tri	-0.41962	0.12792	-3.280	0.001037
riv	0.35248	0.10212	3.452	0.000557
I(riv^2)	-0.34244	0.09419	-3.636	0.000277
rds	-0.12979	0.12776	-1.016	0.309702
agr1:rds	0.53476	0.16150	3.311	0.000929

Table 2. 8 Parameter estimates using the standardized values for variables with coefficients, standard error, Z statistic (standardized normal deviate), and p-value for the best approximating model for wetland presence (including 246 random points and 69 wetland points) for the test data (25% of the full dataset) . Residual deviance for the final model for the training dataset was 279.36 on 307 degrees of freedom (AIC score = 295.36).

	Estimate	Std.Error	z-value	Pr(>|z|)
(Intercept)	-0.74395	0.26160	-2.844	0.00446
agr1	-1.34027	0.30765	-4.356	1.32e-05
trb	-0.61187	0.21682	-2.822	0.00477
tri	-0.67292	0.24122	-2.790	0.00528
riv	0.21289	0.17482	1.218	0.22332
I(riv^2)	-0.07779	0.15984	-0.487	0.62649
rds	-0.14466	0.23523	-0.615	0.53856
agr1:rds	0.38622	0.30627	1.261	0.20730

Table 2. 9 Parameter estimates using the standardized values for variables with coefficients, standard error, Z statistic (standardized normal deviate), and p-value for the best approximating model for wetland presence (including 1000 random points and 265 wetland points) for the full dataset . Residual deviance for the final model for the complete dataset was 1140.1 on 1257 degrees of freedom (AIC score = 1156.1).

	Estimate	Std.Error	z_value	Pr(>|z|)
(Intercept)	-0.58889	0.12756	-4.616	3.90e-06
agr	-1.20067	0.15248	-7.874	3.43e-15
trb	-0.42617	0.09524	-4.475	7.64e-06
tri	-0.47830	0.11247	-4.253	2.11e-05
riv	0.32991	0.08753	3.769	0.000164
I(riv^2)	-0.28103	0.08066	-3.484	0.000493
rds	-0.13803	0.11155	-1.237	0.215958
agr:rds	0.50698	0.14170	3.578	0.000346

Table 2. 10 Chi-square analysis using final logistic regression model of wetland presence on the pre-mined landscape using covariates (agr, trb, tri, riv, rds) plus one quadratic term (riv) and one interaction term (agr:rds).

	Df	Deviance	Resid. Df	Resid Dev	P(>|Chi|)
NULL			1264	1298.59	
agr	1	57.44	1263	1241.14	3.479e-14
trb	1	21.70	1262	1219.44	3.189e-06
tri	1	39.16	1261	1180.29	3.909e-10
riv	1	9.49	1260	1170.79	2.062e-03
I(riv^2)	1	12.72	1259	1158.07	3.622e-04
rds	1	4.71	1258	1153.37	0.03
agr:rds	1	13.24	1257	1140.13	2.736e-04

Table 2. 11 Odds ratio for wetland presence (95% CI) for each variable (agr, trb, tri, riv, riv^2, rds, and agr:rds) included in the final logistic regression model of wetland presence on the pre-mined landscape.

	OR	lower95ci	upper95ci	P value
agr	0.30	0.22	0.41	< 0.001
trb	0.65	0.54	0.79	< 0.001
tri	0.62	0.50	0.77	< 0.001
riv	1.39	1.17	1.65	< 0.001
I(riv^2)	0.76	0.65	0.88	< 0.001
rds	0.87	0.70	1.08	0.216
agr:rds	1.66	1.26	2.19	< 0.001

Table 2. 12 Univariate analysis using agricultural land only (A1) to rank candidate covariates for model approximating wetland presence on the pre-mined landscape. Shown are the parameter estimates with standard error and univariate Wald test statistics. Variables are sorted by residual deviance values (Rdev); smallest to largest.

A1	Estimate	Std.Error	z_value	Pr(>|z|)	Res.Dev
trb	-0.49590	0.13800	-3.593	0.00033	594.83
rdm	-0.29910	0.13480	-2.218	0.02650	604.59
riv	0.23910	0.10390	2.301	0.02140	605.16
rds	0.22690	0.10100	2.247	0.02460	605.50
str	-0.24150	0.12420	-1.944	0.05190	606.23
snk	-14.75230	723.49000	-0.020	0.98400	607.14
slp	-0.19880	0.12730	-1.562	0.11800	607.67
rdl	-0.14740	0.11070	-1.332	0.18300	608.60
tri	-0.14700	0.11470	-1.281	0.20000	608.70
cti	0.10380	0.10310	1.007	0.31400	609.44
elv	0.04757	0.10596	0.449	0.65400	610.24
rdp	-0.04330	0.10640	-0.407	0.68400	610.27
(Intercept)	-1.83070	0.10510	-17.410	<2e-16	610.44

Table 2. 13a Candidate logistic model of agricultural (A1.1) land including two terms: distance to tributary (trb) and distance to surfaced roads (rds). Residual deviance was 588.22 on 757 degrees of freedom (AIC score = 594.22).

	Estimate	Std.Error	z_value	Pr(|z|)
(Intercept)	-1.9464	0.1176	-16.553	<2e-16
trb	-0.5162	0.1397	-3.695	0.00022
rds	-0.3300	0.1399	-2.359	0.01833

Table 2. 14b Candidate logistic model of agricultural (A1.2) land including two terms: distance to tributary (trb) and distance to river (riv). Residual deviance was 589.17 on 757 degrees of freedom (AIC score = 595.17).

	Estimate	Std.Error	z_value	Pr(|z|)
(Intercept)	-1.9334	0.1158	-16.702	<2e-16
trb	-0.5089	0.1394	-3.650	0.000263
riv	0.2490	0.1045	2.382	0.017227

Table 2. 15 Relative comparison of candidate logistic regression models (A1.1, A1.2) for agricultural (A1) land including residual deviance and AIC value.

Model	Covariates	Residual Deviance	AIC
A1.1	TRB + RDM	588.22	594.22
A1.2	TRB + RDS	589.17	595.17

Table 2. 16a Analysis of variance values of chi-square for each term included in the A1.1 logistic regression model (random points = 655, wetland points = 105).

	Df	Deviance	Resid.Df	Resid.Dev	P(>|Chi|)
NULL			759	610.44	
trb	1	15.61	758	594.83	7.8e-05
rds	1	6.61	757	588.22	0.01

Table 2. 17b Analysis of variance values of chi-square for each term included in the A1.2 logistic regression model (random points = 655, wetland points = 105).

	Df	Deviance	Resid.Df	Resid.Dev	P(>|Chi|)
NULL			759	610.44	
trb	1	15.61	758	594.83	7.8e-05
riv	1	5.66	757	589.17	0.02

Table 2. 18 Univariate analysis using non-agricultural land only (A0) to rank candidate covariates for model approximating wetland presence on the pre-mined landscape. Shown are the parameter estimates with standard error and univariate Wald test statistics. Variables are sorted by residual deviance values (Rdev); smallest to largest.

A0	Estimate	Std.Error	z_value	Pr(>|z|)	Res.Dev
slp	-0.85130	0.18620	-4.573	0.00000	598.08
tri	-0.79590	0.16830	-4.729	0.00000	598.59
cti	0.47984	0.09862	4.865	0.00000	605.78
riv	0.34653	0.09709	3.569	0.00036	617.71
rdg	0.33342	0.09378	3.555	0.00038	618.05
elv	0.32761	0.09946	3.294	0.00099	619.46
rds	-0.35055	0.11248	-3.116	0.00183	619.69
trb	-0.28372	0.10864	-2.612	0.00901	623.19
rdp	-0.22818	0.10002	-2.281	0.02250	625.32
str	-0.18642	0.10204	-1.827	0.06770	627.20
rdl	-0.17163	0.10058	-1.706	0.08800	627.68
snk	1.18695	0.91794	1.293	0.19600	629.00
(Intercept)	-0.76837	0.09565	-8.033	0.00000	630.70

Table 2. 19a Model A0.1. Analysis of variance values of chi-square for each term (TRI, CTI) included in non-agricultural land (A0) logistic regression model.

	Df	Deviance	Resid.Df	Resid.Dev	P(|Chi|)
NULL			504	630.70	
tri	1	32.11	503	598.59	1.458e-08
cti	1	4.05	502	594.55	0.04

Table 2. 20b Model A0.2. Analysis of variance values of chi-square for each term (TRI, RIV) included in non-agricultural land (A0) logistic regression model.

	Df	Deviance	Resid.Df	Resid.Dev	P(|Chi|)
NULL			504	630.70	
tri	1	32.11	503	598.59	1.458e-08
riv	1	7.43	502	591.17	0.01

Table 2. 21c Model A0.3. Analysis of variance values of chi-square for each term (TRI, RDG) included in non-agricultural land (A0) logistic regression model.

	Df	Deviance	Resid.Df	Resid.Dev	P(|Chi|)
NULL			504	630.70	
tri	1	32.11	503	598.59	1.458e-08
rdg	1	8.19	502	590.41	4.222e-03

Table 2. 22d Model A0.4. Analysis of variance values of chi-square for each term (TRI, TRB) included in non-agricultural land (A0) logistic regression model.

	Df	Deviance	Resid.Df	Resid.Dev	P(|Chi|)
NULL			504	630.70	
tri	1	32.11	503	598.59	1.458e-08
trb	1	12.66	502	585.93	3.727e-04

Table 2. 23e Model A0.5. Analysis of variance values of chi-square for each term (TRI, RDS) included in non-agricultural land (A0) logistic regression model.

	Df	Deviance	Resid.Df	Resid.Dev	P(|Chi|)
NULL			504	630.70	
tri	1	32.11	503	598.59	1.458e-08
rds	1	6.52	502	592.07	0.01

Table 2. 24 Candidate logistic regression models (A0.1, A0.2, A0.3, A0.4, A0.5) for non-agricultural (A0) land including residual deviance values (R.Dev) and AIC scores.

Model	Covariates	R.Dev	AIC
A1.1	TRI + CTI	594.55	600.55
A1.2	TRI + RIV	591.17	591.17
A1.3	TRI + RDG	590.4	596.41
A1.4	TRI + TRB	585.93	591.93
A1.5	TRI + RDS	592.07	598.07

Table 2. 25 Logistic regression model for agricultural land (A1) with three variables (TRB,TRI, RIV); residual deviance (584.58), degrees of freedom (756) and AIC score (592.58).

Model A1	Estimate	Std. Error	z value	Pr(>|z|)
(Intercept)	-1.454e+00	2.785e-01	-5.221	1.78e-07
trb	-8.486e-04	2.181e-04	-3.891	9.99e-05
tri	-2.042e-01	9.981e-02	-2.046	0.0407
riv	1.323e-04	5.407e-05	2.446	0.0144

Table 2. 26 Logistic regression model for non-agricultural land (A0) including three variables (TRB,TRI, RIV);Residual deviance (582.26), degrees of freedom (501) and AIC score (590.26).

Model A0	Estimate	Std.Error	z_value	Pr(|z|)
(Intercept)	-0.9342	0.1120	-8.341	<2e-16
trb	-0.3418	0.1206	-2.835	0.00459
tri	-0.8677	0.1828	-4.747	2.07e-06
riv	0.2041	0.1067	1.914	0.05565

Table 2. 27 Analysis of variance values of chi-square for each term included in the logistic regression model using data from agricultural land on the pre-mined landscape (A1).

	Df	Deviance	Resid.Df	Resid.Dev	P(>|Chi|)
NULL			759	610.44	
trb	1	15.61	758	594.83	7.8e-05
tri	1	4.27	757	590.57	0.04
riv	1	5.99	756	584.58	0.01

Table 2. 28 Analysis of variance values of chi-square for each term included in the logistic regression model using data from non-agricultural land on the pre-mined landscape (A0).

	Df	Deviance	Resid.Df	Resid.Dev	P(>|Chi|)
NULL			504	630.70	
A0.std$trb	1	7.52	503	623.19	0.01
A0.std$tri	1	37.26	502	585.93	1.036e-09
A0.std$riv	1	3.66	501	582.26	0.06

Chapter 3 Estimating wetland ecosystem intactness

3.1 Introduction

Wetlands at the landscape scale

Globally, wetland loss has been exacerbated by factors including multiple stakeholders with separate agendas for land use, a lack of knowledge regarding wetland function and a lack of consistent government policy to protect wetlands (Turner et al. 2000). The socio-economic value of wetlands has been undermined by a lack of "knowledge and experience with the resource" (Brouwer et al. 1999). Wetlands are considered as natural capital assets with multiple functions (Olewiler 2004; Gren et al. 1994) and are considered integral to post-disturbance restoration (White and Fennesy 2005) and reclamation plans (Chen et al. 2007). Moreover, wetland restoration is increasingly becoming a landscape scale endeavor due to the complexity of interwoven wetland processes that are difficult to separate from the landscape (Simenstad et al. 2006). Recognizing that the spatial location and relative position of wetlands has the potential to affect "functional attributes such as hydrology, shore stability, nutrient supply, sediment-contaminant retention, groundwater recharge, plant community survival, food production export, and wildlife diversity and productivity", (France 2003) emphasized the need for a landscape scale approach to wetland mitigation of any type. In the context of this study, restoration refers to the process of bringing back natural ecosystems (e.g. wetlands) to a prior state with the intent being "reinitiating ecosystem processes" whereas reclamation is used to refer to the process of returning land to a former production system (e.g. agriculture) (Clewell and Aronson 2007). Reclamation is the term more commonly used when referring to overall land use practices and post-mining activities at Genesee.

Wetlands and ecological integrity

The loss, alteration, or degradation of wetlands affects landscape levels of ecological integrity, defined by Forman (1995) as the "combination of near-natural levels of productivity, biodiversity, water, and soil characteristics", and this limits the services that wetlands can provide. Wetland functions provide goods and services such as carbon sequestration (Euliss et al. 2005), biological diversity maintenance (Snodgrass et al.

1999), and wildlife habitat provision (Semlitsch and Bodie 2003). Wetlands are often valued by humans for the services they provide (Mitsch and Gosselink 2000) and many studies have implemented a landscape scale approach to identify landscape patterns and drivers of land use change that affect wetland function (Qui et al. 2009; Moreno-Mateos et al. 2008; Naugle et al. 2001).

Establishing reference conditions

Major challenges in restoring any ecosystem include determining the baseline or reference conditions, the scale of reference, and the time-period prior to disturbance. Historical baselines often refer to the time-period immediately before disturbance as the natural, stable, or even optimal state. Bratton (1992) suggests that this approach is limiting because it creates "a dichotomy not only between natural and human but also between natural and new". Most often, the pre-disturbed landscape, has already been altered from some natural state. Accordingly, the pre-disturbed state should be quantified appropriately and used with caution and caveats. Reference conditions should only be used in conjunction with present day, on-the-ground and site-specific evaluations. Ferrari et al. (2009) found that reclamation after coal mining was not effective for returning the pre-disturbed hydrological regime and that post-mined soils were compacted and impervious, similar to that of urban areas but without proper drainage mechanisms. Nonetheless, reclamation is necessary, and although reference conditions for wetlands may be uncertain because hydrology has been radically altered by mining, these conditions provide a standard against which future conditions can be compared and assessed.

Tools – GIS and Fragstats

Advancements in Geographical Information Systems (GIS) (Padilla 2008; Saito et al. 2007; Domaas 2007) have allowed us to reconstruct past conditions. Several studies have quantified and qualified historical landscapes to determine ecosystem states and guide management decisions for future landscapes (Günlü et al. 2009; Gärtner et al. 2008; Rhemtulla et al 2002). Fragstats, a spatial pattern analysis program for quantifying landscapes (McGarigal and Marks1995) has been used, in conjunction with GIS, to measure landscape structure (Uuemaa et al. 2007; Brownstein et al. 2005).

Brief description of landscape ecology

Landscape structure refers to the composition and configuration of land cover types or patches, which are measurement units described as spatially homogeneous entities; distinct from surrounding lands (Castellon and Sieving 2006; Bender and Fahrig 2005; Goodwin 2003). Composition refers to the number and type of patches on the landscape and configuration refers to patch morphometry, arrangement, and spatial distribution (Baker et al. 2006; Li and Reynolds 1993; McGarigal and Marks 1995). Landscape metrics are used to measure patch structure. Composition metrics are used at the landscape scale to compute representative values of proportion, richness, eveness, and diversity, independent of spatial position or juxtaposition (Leitao et al. 2006; McGarigal and Marks 1995). Configuration metrics are used to characterize individual patches or the neighboring properties of the patches. Overall landscape heterogeneity emanates from the composition and configuration of landscape patches and is described by Lindenmayer and Fischer (2006) as "related to the extent to which a landscape viewed from the air is characterized by a diversity of environmental gradients or patch types".

Landscape metrics

Landscape metrics have been used to help evaluate landscape function and explain the link between landscape process and pattern (Corry et al. 2008; Haines-Young and Chopping 1996). Landscape function refers to the way landscape elements overlap and interact to move energy, materials, and organisms, and it is dependent upon structure (or pattern) (Turner 1989). To distinguish functions associated with pattern, landscapes can be analyzed at three levels: landscape (multiple all-class patches), class (multiple same-class patches), and patch (single homogenous area) (Leitão et al. 2006). Although levels may be interrelated, metrics explain phenomena related specifically, and exclusively, to the elements at that level (Leitão et al. 2006). Understanding what each metric measures and how it measures it, is critical to landscape interpretation and prognosis, and intrinsically linked to the original question or prompt for analysis.

3.1.1 Research Hypothesis

Surface mining alters landscape structure and influences the spatial distribution and abundance of landscape features such as wetlands. I hypothesized that there would be less wetlands and saturated soils, less forested and vegetated areas, and greater fragmentation on the study landscape due to mining operations that require the removal of overburden to get to the coal seam.

3.1.2 Research Objectives

My research objective was to investigate changes in wetland composition, configuration, and overall landscape structure between 1982 (pre-mined) and 2007. To achieve this objective, I needed to detect changes in LULC between two time periods; pre-disturbed (1982) and present-day (2007). The differences between these two dates allowed me to assess the extent to which the reference conditions (1982) should be used for wetland restoration. More specifically I wanted to:

- Determine the effects of cell-size and patch shape and size on metric value output;
- Isolate and describe the spatial variability of wetlands and explain differences between two time periods (1982 and 2007);
- Explain differences in landscape fragmentation between 1982 and 2007.

3.1.3 Research Benefits

My research was designed to address the question of how the relative state of the study landscape differs between 1982 and 2007, and in particular, how the composition and configuration of wetlands changes over time. Results from this study are relevant to a management decision to focus wetland restoration efforts on sites that are considered of the best for supporting and sustaining wetlands rather than on sites selected by historical reference without qualification. With a scientific approach, results from this study should increase the probability of wetland restoration success at the landscape level on a post-mined landscape.

3.2 Study Site

The Genesee Generating Station, owned by Capital power (formerly Epcor) uses coal-fired generators to provide electrical power to Edmonton and the surrounding communities. Coal is provided by an adjacent strip-mine, jointly owned (50/50) by Capital Power and Sherritt Coal. The mine site is located approximately 70km south-west of Edmonton, Alberta (53^{o} 20' 35" N, 114^{o} 18' 17" W) in the North Saskatchewan River Basin (Figure 3.1). The study area falls within the Edmonton Plain sub region of the Eastern Alberta Plains (Pettapiece 1986) and is considered a Dry Mixedwood Sub region of the Boreal Forest Natural Region (Achuff 1994). The landscape is characterized mainly by undulating lowlands and hummocky uplands with clay-textured parent materials, luvisolic and organic soils and an agro-climate of moderate heat limitation (Pettapeice 1986). Summers are short and warm (July average 16.3°C) and winters are long and cold (January average -13.4°C) (Westworth and Brusnyk 1983). For the mine permit area only, local topography and landscape were described by Western Soil and Environmental Services (1981) prior to mining as:

> The region is underlain by a succession of essentially terrigenous sandstones and shales, and coal beds of late Cretaceous and early Tertiary ages. These have been very slightly deformed by regional tectonic and local glacial processes.
>
> Present day uplands reflect preglacial highs to some extent but glacial and alluvial deposits generally subdue bedrock relief. In the Genesee area, the topography varies from gently rolling to flat and slopes generally in a northeasterly to northwesterly direction. Topography ranges from 782m to 730m.

At Genesee, The "upper Ardley coal zone of the Cretaceous-Paleocene Scollard formation is mined for use as thermal coal" (Pollock et al. 2000). The upper Cretaceous and Paleocene stratigraphy consists of the the Paskapoo Formation, Scollard Formation, Battle Formation, Whitemud Formation and Horseshoe Canyon Formation (Pollock et al. 2000).

3.3 Materials and methods

Data overview

I used ArcGIS desktop software (version 9.2) along with the Spatial Analyst extension (Environmental Systems Research Institute Inc [ESRI] 2006). All the data used for this study was projected using a Universal Transverse Mercator Projection (UTM) on the North American Datum of 1983 (NAD83) to match a current (2007) orthoimage (1:30,000) that was georeferenced using ground control points (GCP) from the Genesee mine-site. I used two main data sources from which all the feature classes were derived;

aerial photographs and a Digital Elevation Model (DEM). I used a 1 x 1 m resolution for all raster analysis. Resolution refers to how accurately the map features can be depicted at a given scale and map scale is the extent of reduction expressed as a ratio (e.g. 1:10,000).

3.3.1 Data preparation

3.3.1.1 Source data

Aerial photographs

I obtained pre-mined (1982) black and white aerial photographs (1:10,000) from EPCOR. The photographs were scanned on a flatbed scanner using a resolution of 600 dots per inch (dpi), georeferenced, rectified and mosaiced to create four consecutive map sheets. Photographs that did not overlap with the 2007 orthoimage were georeferenced using 20K base features (scale 1:20,000) obtained from AltaLIS, a not-for-profit agent for Spatial Data Warehouse Limited that maintains Alberta's digital mapping. Exact registration between all photographs was not possible and had to be corrected for during manual delineation and classification of landscape features. I also obtained an orthoimage of the study area (year: 2005; scale 1:30,000; UTM projection: Zone 12, NAD83 datum) to use for a preliminary cell size analysis.

3.3.1.2 Land cover data

To create LULC feature classes, I used editor (sketch tool) in (ArcGIS 9.2, ESRI 2006) to manually digitize all land classes that were visible and identifiable on the orthoimages at a viewing scale of 1:2000 for the entire landscape. Ground-truthing was limited to landscape features that remained relatively unchanged from 1982 to 2007, but I was able to reference relative land classifications using a classified image created in 1998 for biomonitoring purposes by Jacques Whitford and Associates (a consulting firm). AltaLIS base features, including roads and water bodies, were overlaid and used for relative placement and classification but were generally too coarse to be of any direct use. All data feature classes were clipped to the extents of the defined study area and all analyses were conducted using a mask setting in ArcToolBox (ArcMap, ESRI 2006).

Data format conversion

The LULC and hydrological coverages were converted to raster datasets with integer values to retain previously assigned classifications and using a 1-m cell size. Individual rasters were mosaiced together into a single raster dataset. Where rasters overlapped, the order of precedence for assigning cell values was set to hydrology first and LULC second. A raster attribute table was generated for the final raster dataset. The final product included two separate raster datasets, each covering the entire study area; representing two time periods – 1982 and 2007.

3.3.2 Spatial metric selection

Preliminary cell size assessment using 2005 dataset

I used class and landscape level metrics computed with Fragstats 3.3© (McGarigal and Marks1995; herein Fragstats) to quantify both the 1982 and 2007 landscape. To evaluate the effect of cell size on the interpretation of landscape patterns, I first tested a separate dataset from 2005 for the same study area using two different cell sizes (1 and 20-m) to assess differences occurring as a function of raster cell size only. Determining the appropriate cell size before converting vector data to raster data is important because cell size affects the accuracy of the metrics used to measure patch area and perimeter (Theobald 2000). The preliminary assessment study area included the actual study area and surrounding land (11,104 hectares of land). Within the Fragstat parameters, I used the 8-cell rule for patch neighbors instead of the 4-cell rule for this data due to the small patch features of the input raster. Many of the metrics in Fragstats are based on the core area of the patch which is the area within the patch that is further than the specified 'fixed edge depth' distance from the patch perimeter. For the 1-m rasters, 'fixed edge depth' was equal to one and for the 20-m rasters, 'fixed edge depth' was equal to 20. Individual metrics were chosen from the landscape level (Table 3.1) and the program was executed for each raster dataset.

Comparing 1982 and 2007

Class level metrics allow computation of the number of patches and relative landscape representation (% of each land class specified) (McGarigal et al. 2002). Select metrics are more appropriate for qualifying landscape fragmentation (Kamusoko and

Aniya 2007; Koper and Schmiegelow 2006). To describe the spatial attributes related to landscape fragmentation for the two years: (1982 and 2007), six landscape metrics were selected at the class level (Table 3.2); (1) number of patches (NP), (2) patch density (PD), (3) landscape shape index (LSI), (4) mean patch area (AREA_MN), (5) interspersion and juxtaposition index (IJI), and largest patch index (LPI). To compare relative land class compositional changes between 1982 and 2007, I used two class level metrics described in Table 3.2: catchment area (CA) and percent land cover (PLAND). Full descriptions and formulas for each class metric are given in the Fragstats 3.3© user's guide (McGarigal et al. 2002).

3.4 Results

3.4.1 Cell size analysis

Landscape metric values varied with cell size at 1-m or 20-m. Results are herein described for wetlands at the 1 m cell size as W01 and at the 20 m cell size as W20. For forests at the 1 m cell size as F01 and the 20 m cell size as F20. Conversion of polygon features to raster features resulted in 512 - W20 (mean size = 0.28 ha) and 1321 - W01 (mean size = 0.13 ha) for a total difference of 809 wetland patches. There were 187 - F20 (mean size = 10.83 ha), and 212 - F01 (mean size = 9.56 ha) for a total difference of 25 forest patches.

Results for the landscape level metrics for wetlands and forests at the 1 m and the 20 m scale are recorded in Table 3.3 and shown in Figure 3.2 and Figure 3.3 (a-k). Landscape level metric values greater at the 1 m than the 20 m scale for both land features (wetlands, forest) included: Total area (TA), Number of patches (NP), Patch Density (PD), Area-weighted mean shape index (SHAPE_AM), Mean fractal dimension index (FRAC_MN), Cohesion (COHESION), and Aggregation index (AI). Landscape metric values that were greater at the 20 m scale for both land features included: Largest patch index (LPI) and Mean Euclidean nearest neighbor (ENN_MN). One metric, Area-weighted mean shape index (SHAPE_AM), was greater for wetlands at the 1-m scale and greater for forest at the 20 m scale whereas mean core area (CORE_MN) was greater at the 20 m scale for wetlands and greater at the 1 m scale for forest.

3.4.2 Land use change

Land cover

Based on Figure 3.4, total area (ha) increased for natural wetlands and decreased for saturated soils between 1982 and 2007. The percentage of area covered for natural wetlands and saturated soil in 1982 was 0.1 percent for each class for a total of 0.2 percent of the study landscape. In 2007, natural wetlands increased to 0.3 percent and saturated soils remained at 0.1 percent (see Table 3.4). Figure 3.5 shows that forested land and vegetated land cover both decreased from 1982 to 2007. The percentage of land area covered by forest decreased from 20.6 to 11.7 percent, while vegetated areas decreased from 7.2 to 1.7 percent between 1982 and 2007.

Fragmentation analysis

Results of class level metrics for fragmentation analysis are shown in Table 3.5 and Figure 3.6 (a-l). For the forest cover class (FT), there was an increase in NP, PD, IJI and LSI from 1982 to 2007 while there was a decrease in LPI and AREA_MN. For the vegetated cover class (VG), all six metrics (NP, PD, LPI, LSI, AREA_MN, and IJI) decreased from 1982 to 2007. For the natural wetlands cover class (NW), all six metrics (PD, LPI, LSI, AREA_MN, and IJI) increased from 1982 to 2007. For the saturated soils cover class (SS), NP, PD, and LSI decreased from 1982 to 2007 while LPI, AREA_MN, and IJI increased.

3.5 Discussion

3.5.1. Cell size analysis

Changing cell size from 1 m to 20 m had a greater effect on the wetland feature class than the forest feature class. The difference in the total area (ha) of land cover measured was similar between F01 and F20 but noticeably different between W01 and W20. Before raster conversion, wetland polygons ranged in size from 0.0008 ha to 7.02 ha while the forest polygons ranged from 0.034 ha to 225.74 ha. When land cover polygons are converted to rasters, cell attributes are assigned by the class covering the most area, which means that many smaller wetlands will be lost with large cell sizes due to lower representation of space. The conversion of polygons to rasters alters the size, shape, and number of patches for each class and the degree of alteration depends highly on cell size selection (Carver and Brunsdon 1994; Congalton 1997). The loss of "virtual" wetlands (-809) due to selection of cell size in the 'polygon to raster' conversion indicates that the 20-m cell size is inappropriate for the small wetlands found on the study landscape. Differences in forest class values by cell size suggest that information loss is minimal where the patch size is larger than the cell size selected for rasterization. The larger cell size causes a smoothed over effect, causing details, or small features, to be blended in with larger features. Since this research pertains mainly to small features (wetlands), the smoothed over effect that comes with selecting a larger cell size is undesirable. Bettinger et al. (1996) also found that 'operational error' (errors that occur during the manipulation of vector data) was reduced with a smaller grid cell size.

Metric values and biological interpretation

Once the number of patches has been determined by raster conversion, the method of measurement used by each metric affects the output values. Metrics using a cell-center to cell-center measure (such as ENN_MN), are affected by patch size and having a few large patches or many small patches greatly affects the output measurement. When the 20 m cell size was used, the number of forest and wetland patches decreased, and this resulted in increased euclidean distances between same-type patches and accordingly, a lower mean distance (where total distance is divided by total number of patches). The large difference between values with the ENN_MN metric demonstrates the influence of cell size selection on metric values and implies that interpretation of this metric could be misleading; particularly for small patch (wetland) assessment.

The shape (SHAPE_AM) demonstrated that differences occur not only with cell size and feature scale but also with feature type and spatial characteristics. Forest features on the study landscape were generally large and linear whereas wetlands were small, less linear and more 'amoeba' shaped. When cell size was increased to 20 m, wetland shapes were simplified (lost) by the grid which resulted in linear, sometimes square-looking wetlands and a lower metric value for SHAPE_AM. For forest features, the effect of cell size on shape complexity was the opposite. Many small cells replicated a smoother (linear) forest edge whereas fewer larger cells produced a jagged stair-like edge, and this resulted in increased shape complexity

With the mean core area (CORE_MN) metric, I expected to see an increase in core area with increasing patch size and although this occurred for wetlands, the CORE_MN value actually decreased with increasing cell size and increasing patch size for the forest class. The reason for this observation is likely because core uses a pre-defined *edge-distance* that excludes a 'buffer area from edge' from the core measure. I used an edge-distance of 1-m for the 1-m cell calculations and 20-m for the 20-m cell calculations. For forests, where patch number and shape did not change dramatically with a change in cell size, the difference would be less core area due to more area allocated to the *edge-distance*. A true *edge-distance* setting should be justified with biological inference and *a priori* knowledge of edge effects by feature class.

Metric values varied with cell size, metric, patch shape and feature class. Results of the preliminary cell size suggest that the 1 m cell size is appropriate for the forest and wetland classes used for this study where spatial area was limited to 11,000 ha. For larger scale studies (coarser resolution and greater extents), the 1 m cell size would likely prove to be too data intensive for Fragstats.

3.5.2. Land Use change

Wetlands

Variation in the spatial pattern of natural wetlands from 1982 to 2007 was expected but not in the direction that it was detected. The total area of land covered by wetlands more than doubled and the mean patch size of natural wetlands increased with a small increase in the number of natural wetlands. This observation strongly suggests that landscape composition changed from a few small wetlands to few more relatively larger wetlands. Although I had expected to see a decrease in the total number of wetlands due to the change in major land use from agriculture to mining, the opposite was true. The observed increase in wetlands may have been due to detection error on the 1982 landscape where greater forested area may have meant more forested wetlands that were not detected; a common issue with aerial photograph interpretation noted by Tiner (1990). Secondly and similarly, Thibault and Zipperer (1994) detected an "increase in the number of wetlands, total wetland area, mean size and the number of large wetlands (over 5 ha)" where land use had shifted from agricultural to urban and attributed this observation to two occurrences: a decrease in intensity of wetland drainage and a decrease in the intensity of land use. Both occurrences of intensity could be used to explain the observed change in natural wetlands for my study area where agricultural intensity has decreased and wetlands have been allowed to reestablish in areas that were previously cultivated. The observed decrease in presence of saturated soils could also be explained by the decrease of agricultural activity and the increased potential for wetlands to re-establish saturated areas.

An increase in LSI for natural wetlands suggests an increase in shape complexity of wetlands which could be indicative of higher levels of disturbance around the wetlands and fragmentation of wetlands (McGarigal and Marks 1995). Alternatively, LSI may be a poor indicator for wetland patches because "LSI may assign higher values to fragmented but simple patterns than to others with more complex and convoluted shapes" (Saura and Carballal 2004). An increase in LPI and AREA_MN along with a slight increase in NP suggests that the overall increase in wetland land cover may be attributed to a few larger than average wetlands. The pre-mined landscape had 116 natural wetlands with a minimum size of 0.002 ha and a maximum of 0.58 ha whereas the mined landscape had 120 wetlands with a minimum size of 0.002 ha and a maximum size of 3.19 ha (approximately 5.5 times larger than the largest wetland before mining). A few larger wetlands may be a result of water pooling in low topographic areas (where the hydrologic regime is appropriate) and increased overland flow due to decreased interception and uptake by vegetation. An increase in IJI in 2007 suggests that wetlands and saturated soils are more evenly distributed over the mined landscape than the agricultural landscape. This observation supports the theory that wetlands on the pre-mined landscape were drained away from agricultural land (major land use) and into non-agricultural land (less than 10% of the landscape); thus producing a clumped or uneven pattern of wetlands. Forest patches demonstrated a decrease in IJI and this suggests that mining has produced a more uneven distribution of forest patches.

Vegetation

The total loss of vegetated and forested land area is indicative of land clearing for mining. The increase in the NP and PD of forested patches along with a decrease in the LPI and AREA_MN indicates increased forest fragmentation. The LSI for forests increased only slightly indicating some disaggregation and an affiliated increase in landscape complexity. The slight increase in IJI suggests that the remaining forested patches are more even or more aggregated. The decrease in PD, LPI, AREA_MN, and

IJI of vegetated patches is relative to the drastic decrease in the NP and indicates that vegetated patches are on average; smaller, sparser, less aggregated, more uniform in shape, and overall, rare on the 2007 landscape. Combined, the class-level metrics for forested and vegetated areas suggest an increase in landscape fragmentation from 1982 to 2007. Visual inspection of the land cover maps for 1982 and 2007 further suggest that the degree of change in class-level metric values is highly variable due to the pattern of land clearing at the core of the study area.

3.6 Conclusions

The Genesee landscape has undergone extreme land cover change as a result of coal mining activity. Forests, shrubs, grasslands, and willows have decreased considerably in the last 25 years since mining was initiated because the majority of this land has been converted into cleared areas of land for various mining purposes (access, storage, buildings). Much of the permitted land is not, and has not been to date, actively mined and therefore it is less intensively managed than alternative land uses such as agriculture – the dominant land use prior to mining. As a result, wetlands may be re-occurring in areas where they were once drained for agriculture.

Landscape metrics can measure landscape patterns and in particular, the composition and configuration of the patches that make up the pattern. The metrics that I examined were sensitive to cell-size selection and patch type; a determinant of mean patch size and shape. The effect of cell size on metric output was greater for small landscape features such as wetlands, which was likely due to loss of information during the conversion from vector (polygon) to raster. Similarly, Hargis et al. (1998) argued that landscape metrics are only useful when the user understands the range of attainable values and the limitations of each measure, and is aware that metric values will vary by patch type. For small wetland features (0.002 ha), metrics that measure shape complexity (e.g. SHAPE_AM) are limited by the cell size of the data (even at 1 m) because the feature shape is oversimplified. Nevertheless, small wetlands are important (particularly as habitat for amphibians) and though spatial resolution is a limiting factor in accurately measuring shape-specific parameters, there are other more suitable metrics (e.g. NP) that can assist wetland evaluation at this scale.

Results of my study demonstrate that forests, vegetated areas, and saturated soils have decreased while natural wetlands have increased and that overall, land cover changes have resulted in increased landscape fragmentation. Thus, all *a priori* predictions were supported except natural wetlands increased instead of decreased. The replacement of forests and vegetated areas from this landscape with cleared land and the increase in fragmentation indicates a reduction of landscape quality. Mining activities may cause impermeable soils due to compaction (Simmons et al. 2008) along with an altered water table and these qualities contribute to the reduction of landscape quality. The main reason for this research, however, was to qualify the pre-mined landscape as a wetland restoration reference and although the pre-mined landscape was, relative to the present landscape less fragmented, it was not wetland-friendly due to the dominant land use being agriculture. An observed increase in natural wetlands on the present day mine site should not suggest that restoration efforts are unnecessary but rather that 1) the potential for this landscape to sustain wetlands may be greater than what has been observed so far and 2) restoration efforts should focus on the landscape as a whole, and in particular, reforestation and overall heterogeneity.

A directive towards a heterogeneous landscape, replete with large forest patches and wetlands is not only feasible, but arguably more suitable for this landscape than replicating the pre-mined agricultural landscape. Although reclaiming mined land to agricultural land is the primary management objective, it could be improved by the natural services that forests and wetlands can provide such as erosion control, nutrient transport and filtering, and ground water recharge. Barrett (1992) addresses issues of agroecosystem ecology by coining the term "agrolandscape ecology" to address 1) agricultural lands at the landscape scale, 2) the interaction between agricultural land and natural systems, and 3) landscape design and management based on sustainability. With advances in remote sensing and GIS, this concept of agrolandscape ecology should not only be easier, but mandatory for future landscapes. Inferences from this study could be used for coarse scale reclamation, restoration, or remediation plans to design sustainable agricultural landscapes where reference conditions are available for assessment.

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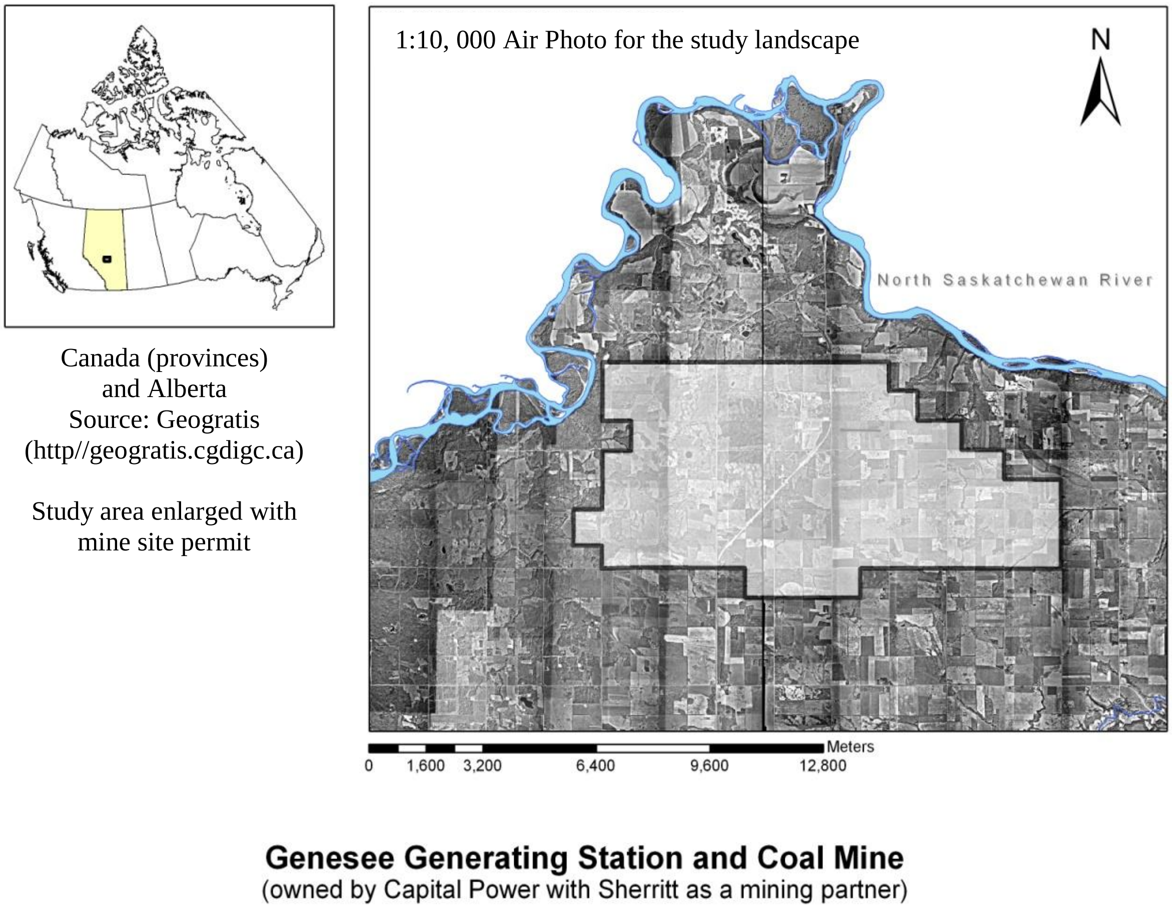

Figure 3. 1 Image showing extents of the study extents belonging to the present-day mine permitted area (shaded box outlined with black) on top of the historical (1982) study landscape. The Genesee Generating Station and coal mine site is located approximately 70km south west of Edmonton, Alberta Canada.

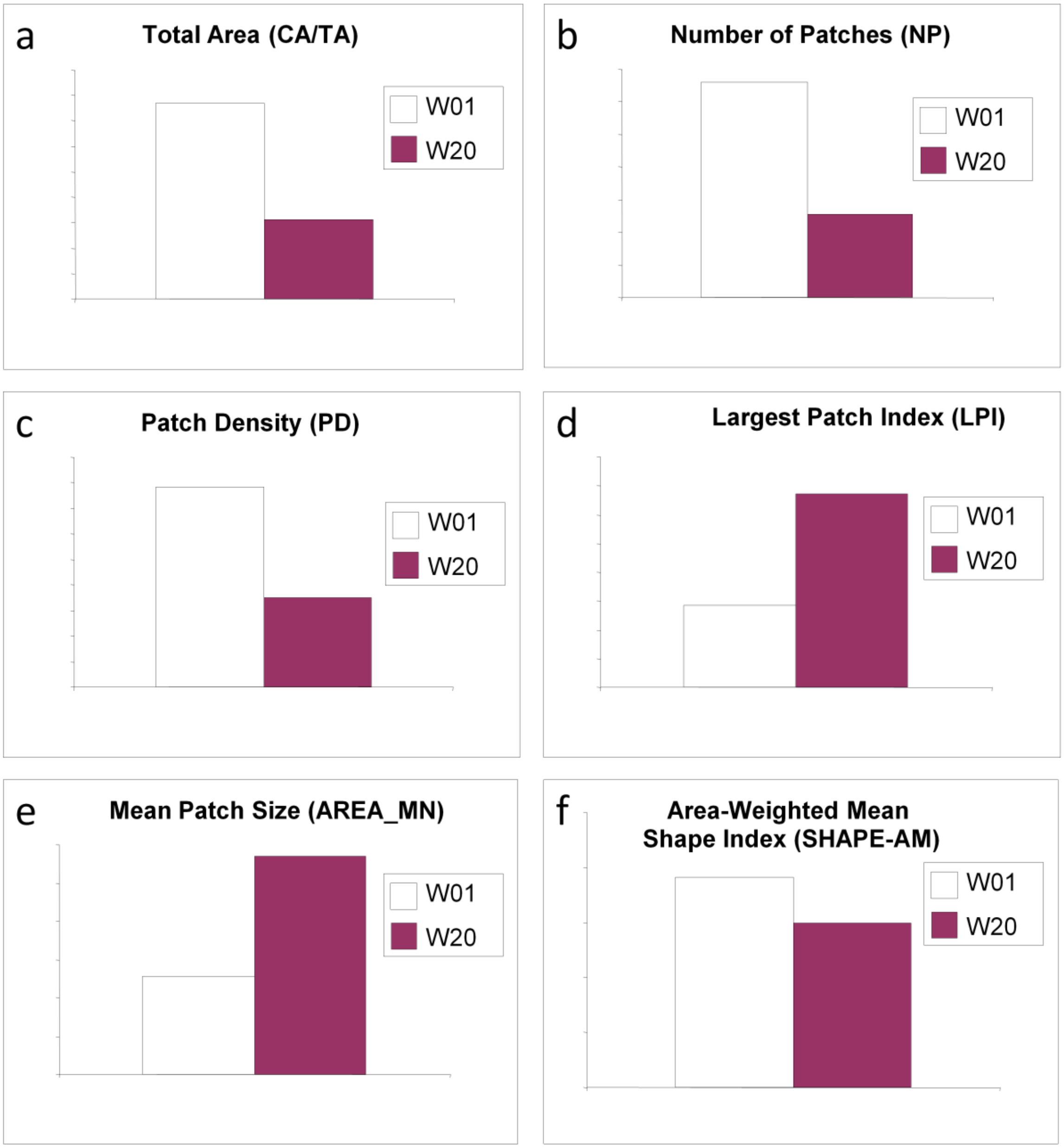

Figure 3. 2 Bar plots showing results of the landscape metrics used to measure wetland patches on the 2005 dataset; comparison of cell size equal to 1-m (W01) or 20-m (W20). The x-axis represents W01 (white bar) and W20 (solid bar) and the y-axis represents the metric value. There 11 metrics include: a) TA, b) NP, c) PD, d) LPI, e) AREA_MN, f) SHAPE_MN, g) FRAC_MN, h) CORE_MN, i) ENN_MN, j) COHESION, k) AI. Metrics g-k are shown on the next page.

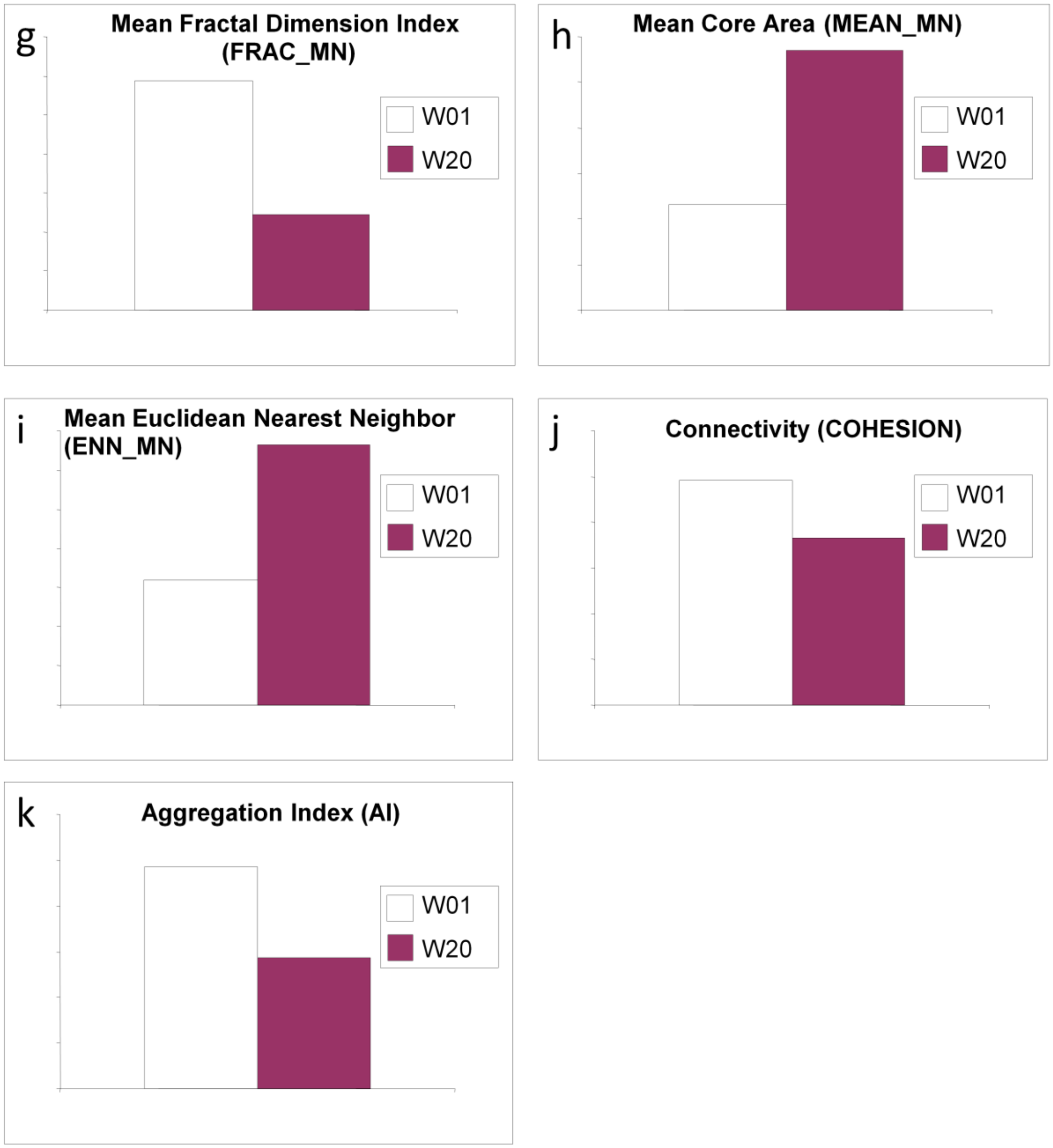

Figure 3.2 continued. Barplots showing results of the landscape metrics used to measure wetland patches on the 2005 dataset; comparison of cell size equal to 1-m (W01) or 20-m (W20). The x-axis represents W01 (white bar) and W20 (solid bar) and the y-axis represents the metric value. There 11 metrics include: a) TA, b) NP, c) PD, d) LPI, e) AREA_MN, f) SHAPE_MN, g) FRAC_MN, h) CORE_MN, i) ENN_MN, j) COHESION, k) AI.

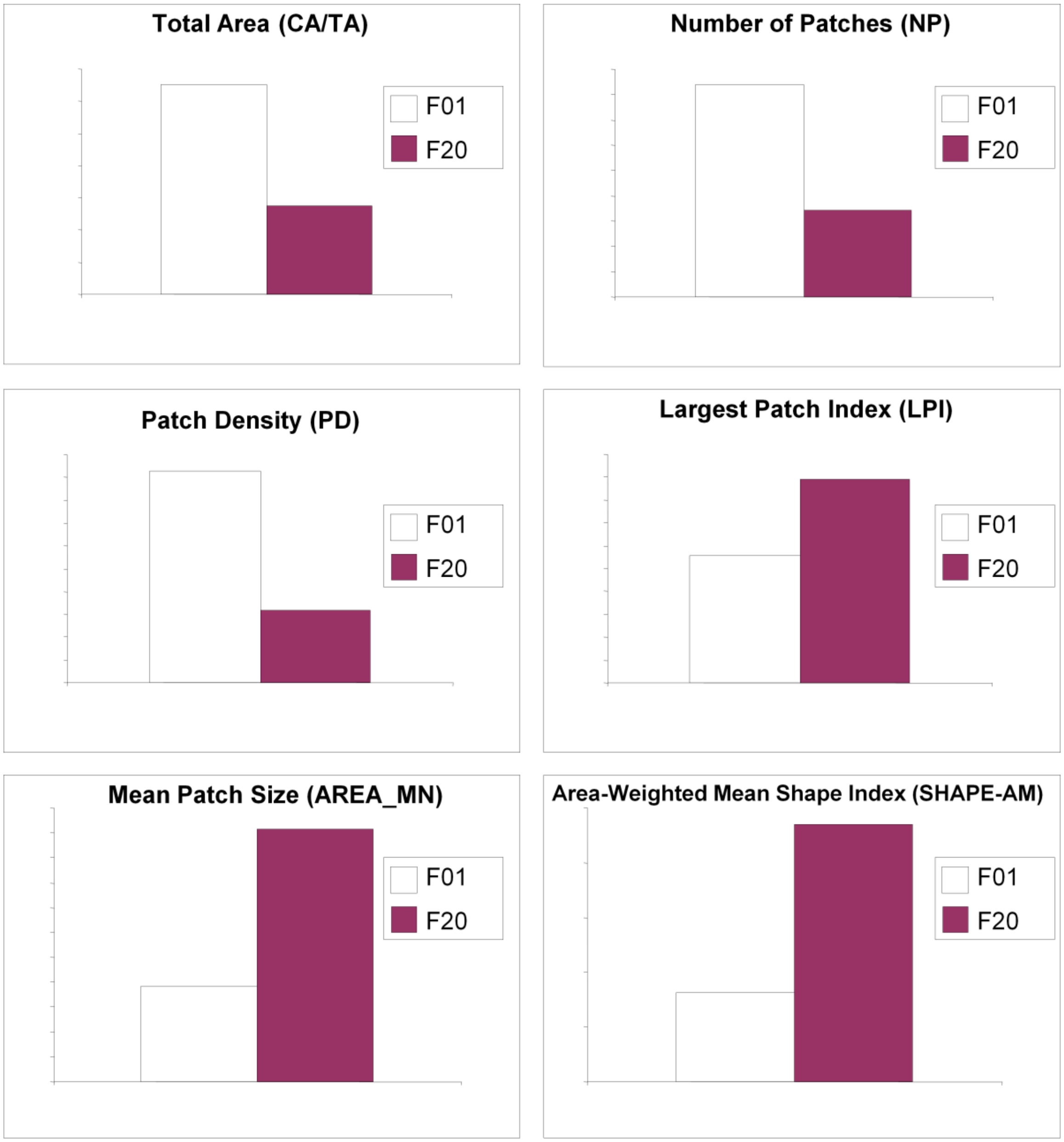

Figure 3. 3 Bar plots showing results of the landscape metrics used to measure forest patches on the 2005 dataset; comparison of cell size equal to 1-m (F01) or 20-m (F20). The x-axis represents F01 (white bar) and F20 (solid bar) and the y-axis represents the metric value. There 11 metrics include: a) TA, b) NP, c) PD, d) LPI, e) AREA_MN, f) SHAPE_MN, g) FRAC_MN, h) CORE_MN, i) ENN_MN, j) COHESION, k) AI. Metrics g-k are shown on the next page.

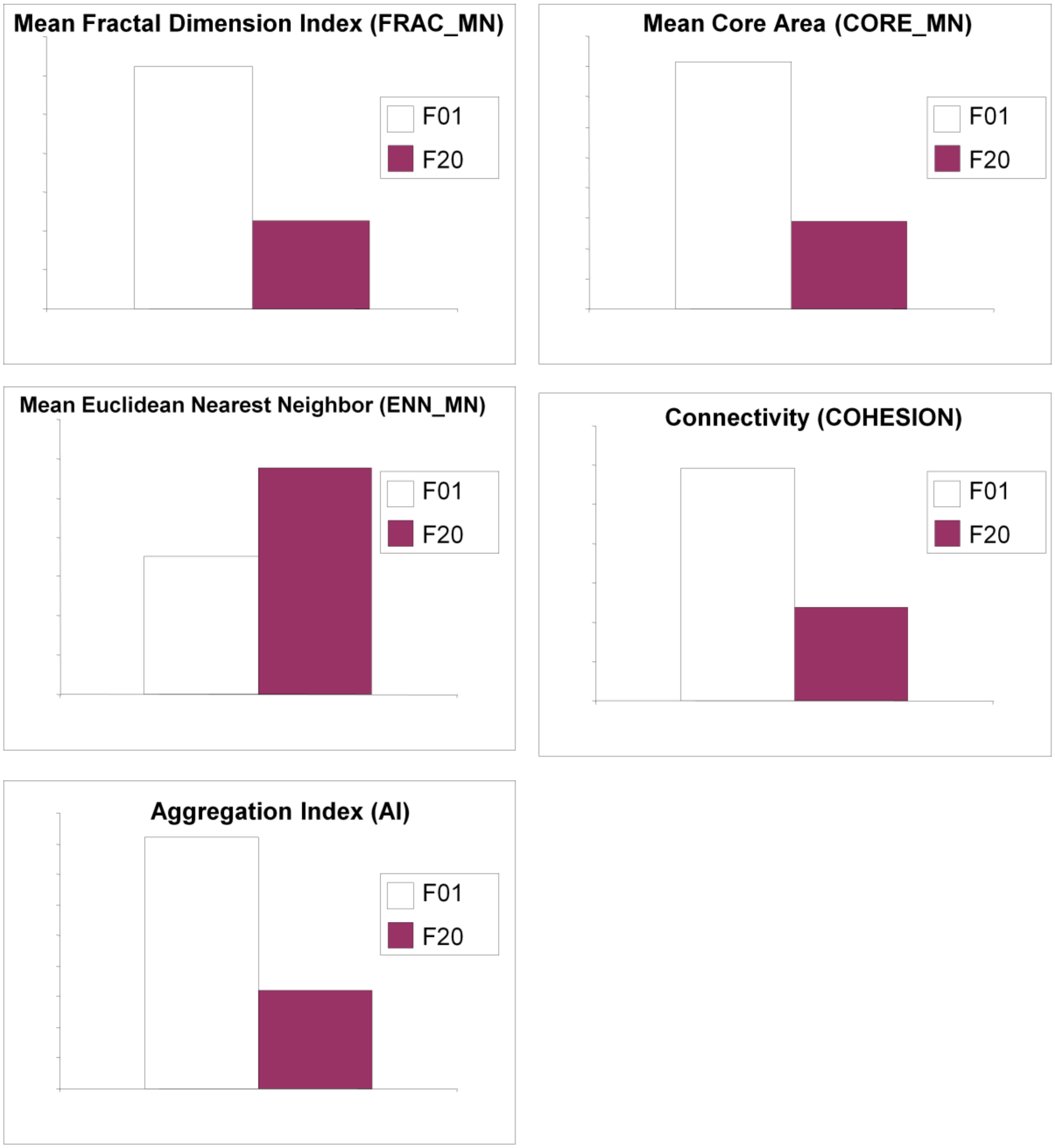

Fig. 3.3 continued. Barplots showing results of the landscape metrics used to measure forest patches on the 2005 dataset; comparison of cell size equal to 1-m (F01) or 20-m (F20). The x-axis represents F01 (white bar) and F20 (solid bar) and the y-axis represents the metric value. There 11 metrics include: a) TA, b) NP, c) PD, d) LPI, e) AREA_MN, f) SHAPE_MN, g) FRAC_MN, h) CORE_MN, i) ENN_MN, j) COHESION, k) AI.

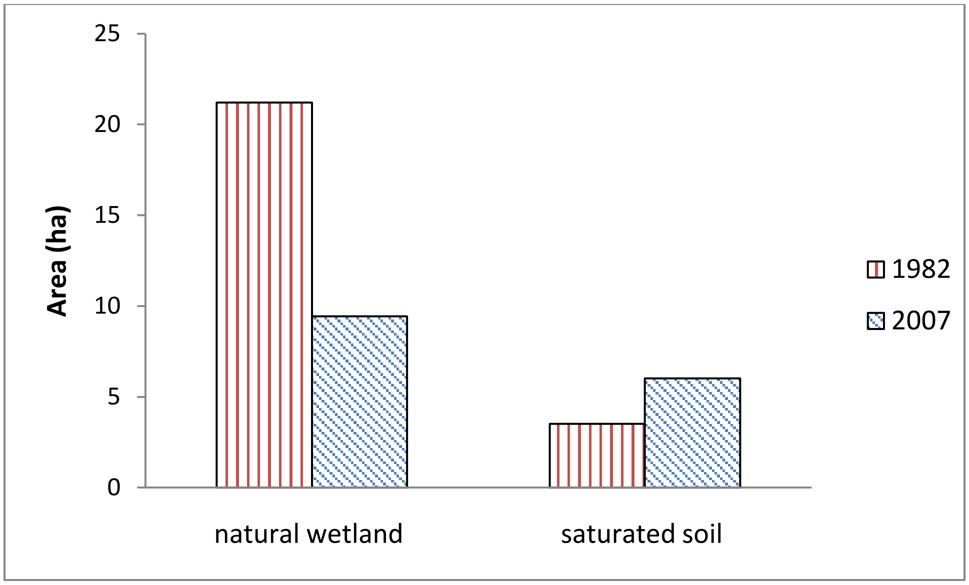

Figure 3. 4 Barplot showing the change in total area covered (ha) for two land cover classes: natural wetlands and saturated soil for the years 1982 and 2007.

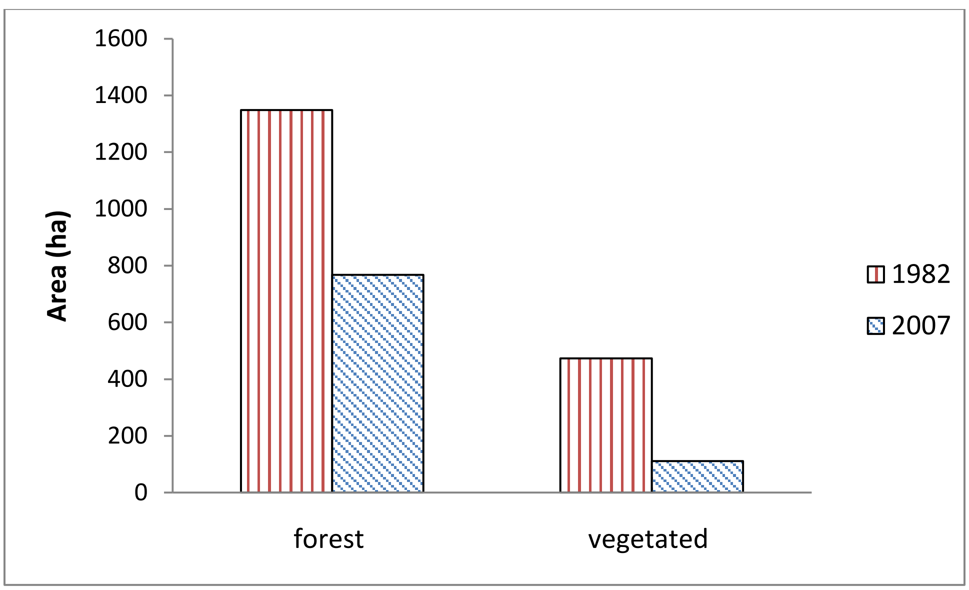

Figure 3. 5 Bar plot showing the change in total area (ha) for two land cover classes: forest and vegetated for the years 1982 and 2007.

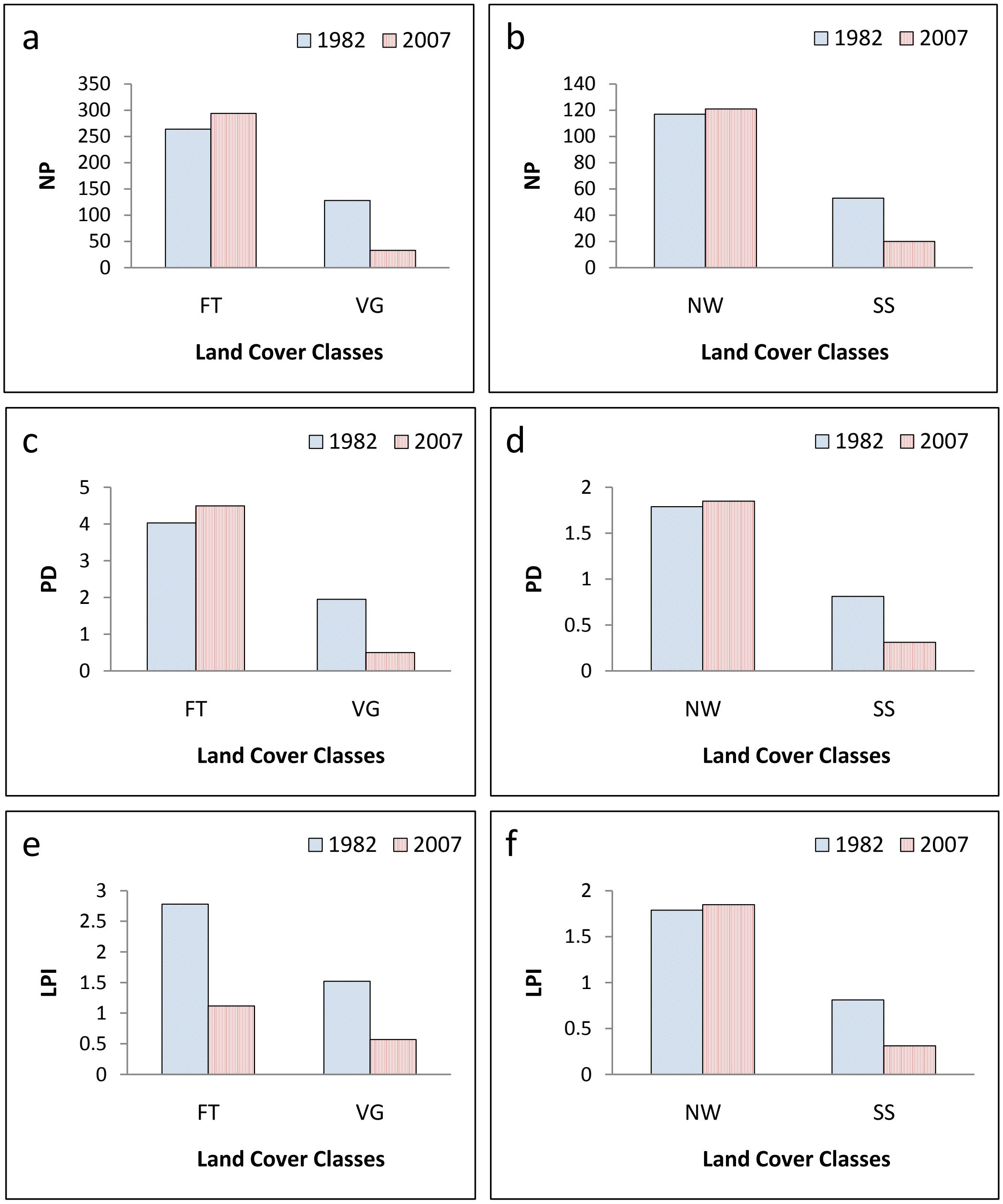

Figure 3. 6 Bar plots showing the change in land cover classes (forested [FT], vegetated [VG], natural wetlands [NW], saturated soil [SS]) for each of six metrics (a/b: number of patches NP; c/d: patch density; e/f: landscape shape index [LSI]; g/h: landscape shape index [LSI]; i/j: area mean value [AREA_MN]; k/l: interspersion and juxtaposition index [IJI])used to assess and compare degree of landscape fragmentation between 1982 and 2007. Continued on next page:

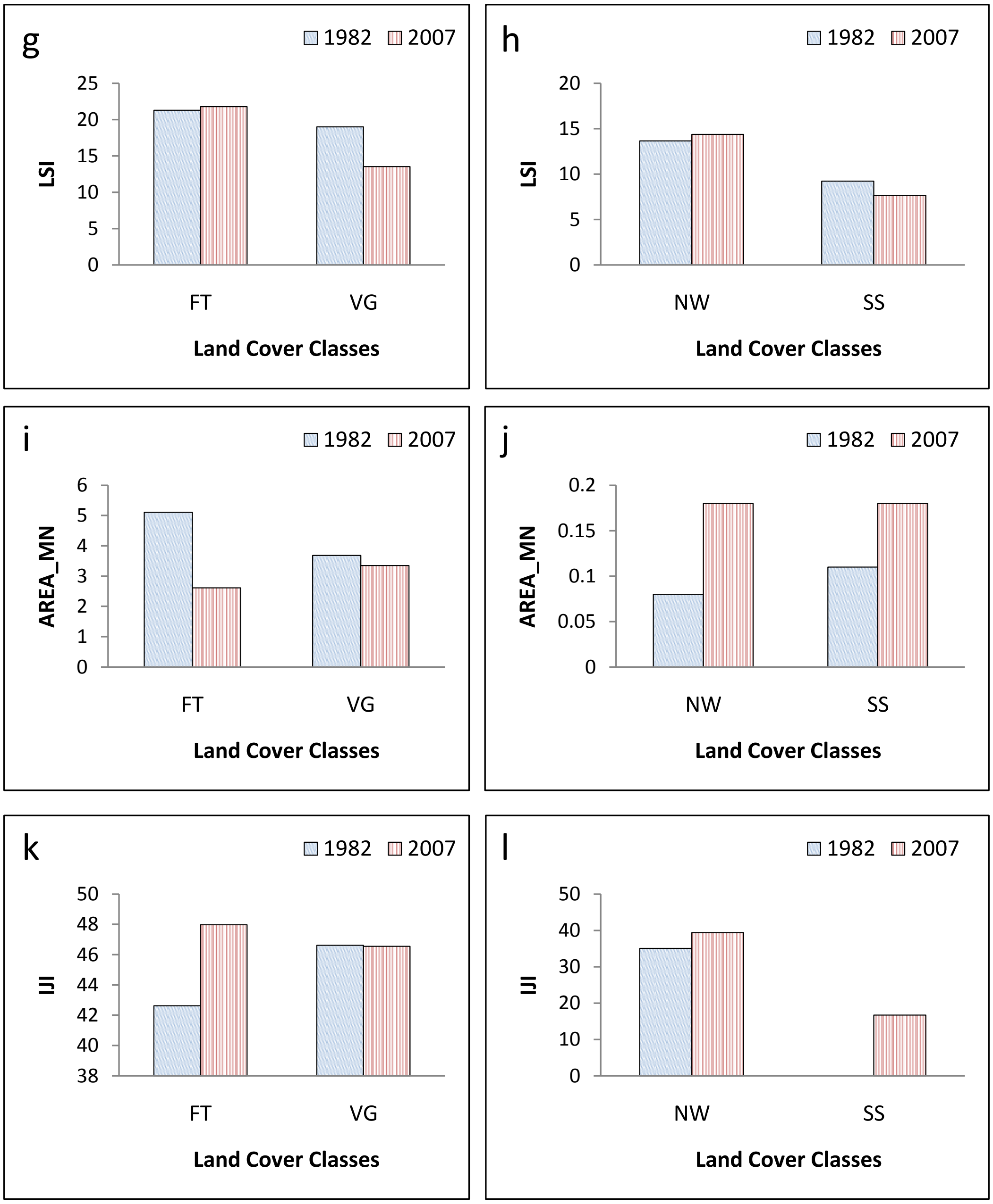

Figure 3.6 continued. Bar plots showing the change in land cover classes (forested [FT], vegetated [VG], natural wetlands [NW], saturated soil [SS]) for each of six metrics (a/b: number of patches NP; c/d: patch density; e/f: landscape shape index [LSI]; g/h: landscape shape index [LSI]; i/j: area mean value [AREA_MN]; k/l: interspersion and juxtaposition index [IJI])used to assess and compare degree of landscape fragmentation between 1982 and 2007. Continued on next page:

Table 3. 1 Description of landscape level metrics (Fragstats 3.3) used for preliminary assessment of raster cell size using the 2005 dataset. Table continues on the following page.

	LANDSCAPE Metric	*ACRONYM*	*DESCRIPTION*	*UNIT OF MEASURE*	*LIMITS*
a	Total landscape area	TA	Total area of landscape (ha)	Hectares	TA > 0…
b	Number of patches	NP	calculates the total number of patches in the landscape	None	NP≥1, No limit
c	Patch Density	PD	measures the number of patches relative to the area of the landscape	#/100 ha	PD>0, Constrained by cell size
d	Largest patch index	LPI	Calculates what percentage of the land is covered by the largest patch (%)	Percent (%)	0<LPI≤100
e	Area mean	AREA_MN	calculates the mean area of patches for the entire landscape	Hectares	AREA>0, No limit
f	Shape area-weighted mean	SHAPE_AM	calculates the sum of shape values for all patches and multiplies it by the proportional abundance of the patches	None	SHAPE≥1, No limit
g	Fractal dimension index mean:	FRAC_MN	Measures the average complexity of edge or perimeter for all patches on the landscape	None	1≤FRAC≤2
h	Core area	CORE-MN	Core area: calculates the mean core area of patches for the entire landscape	Hectares	CORE≥0, No limit
i	Euclidean nearest-neighbor distance mean	ENN_MN	Calculates the average distance of all patches to the same patch classes for the landscape		ENN>0, No limit

Table 3.1 continued. Description of landscape level metrics (Fragstats 3.3) used for preliminary assessment of raster cell size using the 2005 dataset.

	LANDSCAPE Metric	*ACRONYM*	*DESCRIPTION*	*UNIT OF MEASURE*	*LIMITS*
j	Connectivity	COHESION	Measures the corresponding connectiveness of patch types; as cohesion decreases, fragmentation increases.	None	0≤COHESION≤100
k	Aggregation Index:	AI	Similar to contagion, measures an area-weighted mean class aggregation so that each class is weighted by its percentage of land cover		

Table 3. 2 Description of class-level metrics (Fragstats 3.3) used to describe the spatial attributes of land classes and comparing fragmentation and composition between 1982 and 2007.

	LANDSCAPE Metric	*ACRONYM*	*DESCRIPTION*	*UNIT OF MEASURE*	*LIMITS*
Fragmentation	Number of patches	NP	Total number of patches in each class	None	NP ≥ 1 no limit
	Patch density	PD	Number of patches identified by class per unit area	(units*$(100)^{-1}$)	PD > 0, constrained
	Landscape shape index	LSI	Measures aggregation or clumpiness of class type; increasing value corresponds with increasing clumpiness.	None	LSI ≥ 1, no limit
	Area mean value	AREA_MN	Calculates mean area of patches classified by class type.	Hectares (ha)	AREA > 0, no limit
	Interspersion and juxtaposition index	IJI	Measure the juxtapositioning of a patch type with all other patch types.	Percent (%)	0 < IJI ≤ 100
	Landscape shape index	LPI	Percentage of land covered by the largest patch in each class (%)	Percent (%)	0 ≤ LPI ≤ 100
composition	Total (class) area	CA	Total class area	Hectares (ha)	CA > 0, no limit
	Percentage of landscape	PLAND	Ratio of wetlands on the landscape (%)	Percent (%)	0 ≤ PLAND ≤ 100

Table 3.3 Table showing comparison of landscape metric values (Fragstats 3.3) for the wetland and forest land classes using 1 and 20-m raster cell sizes.

LANDSCAPE MEASURE	METRIC	WETLANDS		FOREST	
		1-m cell	**20-m cell**	**1- m cell**	**20-m cell**
Total Area (hectares)	TA	168.35	145.52	2026.77	2024.88
Number of patches (#)	NP	1321	512	212	187
Patch Density	PD	784.68	351.84	10.46	9.24
Largest Patch Index	LPI	4.178	4.95	11.14	17.81
Mean Patch size	AREA_MN	0.13	0.28	9.56	10.83
Area-weighted Mean Shape Index	SHAPE_AM	1.92	1.49	2.13	2.29
Mean Fractal Dimension Index	FRAC_MN	1.12	1.05	1.12	1.08
Mean Core Area	CORE_MN	0.12	0.28	9.44	8.38
Mean Euclidean Nearest Neighbor	ENN_MN	63.72	132.97	70.56	115.28
Connectivity	COHESION	98.31	73.23	99.61	97.11
Aggregation Index	AI	96.68	57.39	99.83	92.76

Table 3.4 Table of land cover classes quantified by total area (ha), percent land cover (%), percent (%) change in area, and change in the number (No.) of patches for two study time periods (1982, 2007).

	1982		**2007**		**1982-2007**	**1982-2007**
Land cover type	*Area (ha)*	*%*	*Area (ha)*	*%*	*% change*	*No. patch change*
forest	1348.27	20.6	766.75	11.7	-8.9	30
vegetated	472.45	7.2	110.60	1.7	-5.5	-95
natural wetland	9.44	0.1	21.20	0.3	0.2	4
saturated soil	6.02	0.1	3.51	0.1	0.0	-33

Table 3.5 Table showing class level metric results for fragmentation analysis for 1982 and 2007, for four land cover classes (FT, VG, NW, SS) and six metrics (NP, PD, LPI, LSI, AREA_MN, IJI).

		NP		PD		LPI	
Land cover class	**TYPE**	**1982**	**2007**	**1982**	**2007**	**1982**	**2007**
forest	FT	264	294	4.03	4.49	2.78	1.12
vegetated	VG	128	33	1.95	0.50	1.52	0.57
natural wetland	NW	117	121	1.79	1.85	0.02	0.05
Saturated soil	SS	53	20	0.81	0.31	0.01	0.03

		LSI		AREA_MN		IJI	
Land cover class	**TYPE**	**1982**	**2007**	**1982**	**2007**	**1982**	**2007**
forest	FT	21.30	21.77	5.11	2.61	42.62	47.97
vegetated	VG	18.99	13.55	3.69	3.35	46.62	46.55
natural wetland	NW	13.66	14.36	0.08	0.18	35.06	39.46
Saturated soil	SS	9.24	7.66	0.11	0.18	0.00	16.74

Chapter 4 Summary, Implications, Limitations and Recommendations

4.1 Research Summary

Defining Equivalent capability

Equivalent capability is an Alberta-specific term adopted by Alberta's EPEA (Province of Alberta 1992) and used by Alberta-based companies (e.g. Suncor, Petro-Canada, Cumulative environmental Management Association (CEMA), Laricina Energy Ltd., and Conoco Phillips) in various reclamation-related documents. Without knowing the initial state or the designated purpose of a specific land cover, "equivalent capability" is value-laden. Smyth and Dearden (1998) suggest that this approach "lacks site-specificity and lacks quantitative measurements" and does not address post-reclamation ecosystem functioning. Moreover, Smyth and Dearden (1998) recommend that Alberta jurisdictions adopt a functional approach to post-mining reclamation by "involving landscape, ecosystem, community and population level monitoring". My research approach helps define and specify conditions at the landscape level for which "equivalent capability" can become meaningful and useful.

Assessing landscape capability

The ultimate goal of this research was to assist reclamation efforts to build sustainable wetlands, by defining reference conditions for the Genesee coal mine.. In chapter two I examined a range of potential anthropogenic and natural variables related to wetland presence on the pre-mined landscape. In chapter three I described changes in landscape spatial pattern between 1982 and 2007. The premise for both chapters was that landscape structure affects wetland function. Mitsch and Gosselink (2000) argue that wetlands are valued by humans for the functions they provide but ultimately, these functions are related to landscape structure and affected by hydrogeomorphic position. Bedford (1996), explains "hydrologic equivalence" by "the kinds, numbers, relative abundances, and spatial distribution of wetland templates" and the importance of these landscape characteristics in providing information regarding "if and where a replacement wetland is likely to attain long-term hydrologic equivalence with the lost wetland". Results of my research indicate that pre-mined wetland presence was influenced by agriculture land use more so than by natural features. I also show that landscape change from agriculture to mining has resulted in more surface water and fragmented land cover as a result of deforestation. An altered hydrologic regime due to land clearing, mainly deforestation, has been observed by other researchers in Alberta's Beaver River basin (Fennell et al. 2006), Alberta's boreal forest (McNabb et al. 2001), and in the Continental United States (Greene et al. 1999). Going forward from these results, I discuss the implications of my results for selecting suitable future sites for wetlands at Genesee.

4.2 Research Implications

Results from my research may have important implications for wetland restoration success at the landscape scale using a quantified historical state of reference. First, Building a pre-mined landscape inventory including wetland composition (number, type, and abundance) and wetland configuration (spatial arrangement) of wetlands allows future land use plans to be developed with a defined historical reference state to which all future states can be compared and evaluated. Second, Understanding the processes (e.g. agricultural land conversion) that have created landscape patterns observed in the reference state helps managers prevent or ameliorate similar undesirable patterns on future sites. Third, recognizing the capacity or untapped potential of the landscape to sustain wetlands alongside with agricultural and alternative land uses increases the probability of maximizing value and desirable wetland function on a reclaimed landscape.

Methods used for this project could be used for any landscape where aerial photographs, satellite images, or even photographs are available to re-construct reference states. The greatest benefit of a landscape scale assessment for wetland restoration is the potential to reduce narrow the list of potential restoration sites prior to field reconnaissance. For managers, assessing wetlands at the landscape scale makes it possible to predict cumulative effects of land use and land cover on wetlands at a broad scale. Lastly, this study is a case study showing the potential to monitor changes in land use and cover over time and space, therefore showcasing a great tool for land stewardship.

4.3 Research Limitations

For the scale of this study it was feasible to scan, georeference and tile together numerous 1:10 000 aerial photographs to reconstruct the past and to delineate multiple, tiny wetlands using a 1 m cell size. The minimum mapping unit (e.g. smallest wetland detected) and cell size are limited by data processing power and would need to be adjusted accordingly for larger study areas. Reconstructing past landscapes is limited by image type (satellite versus aerial), scale, and availability by year and by coverage. Accurate wetland detection on aerial photographs depends on the photo-interpreter and is therefore subject to error, particularly for historical photographs where ground-truthing is not possible. For my study, forested wetlands may not have been detected due to canopy cover and ephemeral wetlands may not have been detected due to dry periods. Hard-copy aerial photographs were used to compare inter-year images where forest cover differed. Landscape analysis is limited to broad generalizations about the observed patterns and should not be used as the sole reference for quantitative and qualitative data. Stromquist et al. (2000) recommend that on-the-ground, site-specific fieldwork and an interdisciplinary approach at the local level are necessary for landscape level analyses where the aim is to understand the underlying processes of change. Lastly, my research was limited by the availability of soils and geologic data for the pre-mined landscape. Detailed soil and hydrogeomorphic characteristics of the landscape may further explain patterns of wetland presence and explain the relationship between wetland morphology and function (Shaffer et al. 1999).

4.4. Research recommendations

4.4.1 General wetland design considerations

Mitsch and Wilson (1996) suggest three ways to improve the success of wetland creation and restoration: (1) Understand wetland function; (2) Give the system time; and (3) Allow for the self-designing capacity of nature. Before wetlands can be created or restored, a detailed understanding of wetland “plants, soils, wildlife, hydrology, water quality, and engineering” is necessary (Mitsch and Wilson 1996). Results from my research could be used to select future wetland sites most likely to enhance desired wetland function. Wetlands on an agricultural landscape, for example, can reduce agrochemical contamination of surface and groundwater but this function depends on wetland siting, design and landscape position (Crumpton 2001). Once a wetland is created, wetland functions may require 15-20 years to fully develop (Mitsch and Wilson 1996). The Landscape scale approach used for my research could help to identify wetland sites where surrounding land uses are known and potential and future land cover changes can be prescribed or identified. More specifically, land could be allocated to wetlands based on proximity to forest or vegetated cover, and evaluated by the potential for surrounding land to be cleared or re-vegetated. Lastly, Mitsch and Wilson (1996) recognize alternative approaches to wetland design: designer and self-design. The designer approach was implemented at my study site (Genesee) by a former graduate student (Shifflett 2005) who transplanted donor soil from local wetlands (slated for disturbance) to an area on the mine-site flooded to be a wetland. Due to the success of community composition in her test plot, Shifflett concluded this to be a viable alternative for Genesee. She also observed colonization in nearby opportunistic wetlands and suggested that the self-design method would also be feasible for wetland creation. Selecting potential wetland sites at the landscape scale could be used to determine where each method of wetland creation (designer or self-design) would be more appropriate. For example, wetland sites down-slope from established wetlands where nutrient and seed transportation between wetlands is likely, may allow for a self-design approach. Conversely, wetlands isolated by distance or surrounding land use could be identified for potential intervention, where donor soil and plant propagules are more appropriate for establishing wetland plant communities.

4.4.2 Wetland sites for Genesee

Reclamation practitioners at Genesee should assume that the following landscape attributes have been altered by mining and may continue to be affected long after mining is completed: 1) Hydrology and soils, 2) topography 3) land cover and land use. However, with foresight and planning, mining holds the potential to configure landscape to facilitate wetland presence.

Hydrology and soils

One of the greatest challenges of restoring or creating wetlands on the Genesee landscape will be identifying the hydrogeomorphic regime on reclaimed land. Bonta et al. (1992) evaluated effects of coal mining on physical conditions and ground-water hydrology and observed altered states of sub-surface flow paths, ground-water levels and hydraulic conductivity. Results of their study varied by site and although all sites demonstrated extreme changes in hydrological conditions, the extent and direction of change were complicated by local conditions and mining-specific impacts. Local conditions include the physical and chemical condition and profile development of reclaimed soils that affect the potential for wetland restoration. The removal, replacement, redistribution and grading of sub soil and top soil after mining at Genesee will require time for settlement and development of drainage patterns before wetlands can be permanently established.

Topography

Topography (slope and elevation) are altered by mining but the potential exists to either mimic the pre-mining topography or modify the reclaimed topography. The pre-mined landscape was approximately 70% agronomic land use which correlates with cleared, homogenous, drained and flat land locations. Reclamation opportunities exist to tailor a percentage of the land for wetlands (e.g. create land depressions). Using a wetland suitability map, potential restoration and wetland creation sites can be identified prior to reclamation and designed to accommodate wetlands with appropriate grading.

Land cover and land use

Along with the early objectives to reclaim the Genesee landscape to prime agricultural land, managers have added a new objective to “re-establish wetlands” (Capital Power Corporation 2009), and to incorporate natural features back into a landscape that was previously disturbed by agricultural land conversion. Putting wetlands back amongst agricultural land should be an exercise of selecting appropriate sites based on the overall composition and configuration of other wetlands and land uses. A landscape scale site plan will help to intersperse wetlands with alternate land uses and achieve a pattern of distribution that is conducive to landscape level wetland functions.

4.5 Recommendations for wetland planning and management

I have several recommendations for wetland planning and management at Genesee:

- Incorporate soils data from biomonitoring (or in-house surveys) and create a GIS feature class of estimated soil zones for the current mine site. Soil data is pertinent because "hydric soil morphology and genesis relate important information about the nature of wetland hydrology" (Richardson et al. 2001), and wetland success is related to a wetland's hydroperiod (Gamble and Mitsch 2009).
- Verify mapping accuracy of land class features such as wetlands, forests, vegetation, and cleared areas; particularly where land cover has changed since 2007. Reliable inferences about ecological processes from landscape patterns depends highly on mapping accuracy and flawed information may result in poor management decisions for land use planners (Shao and Wu 2008).
- Qualify wetlands with on-the-ground assessments using either the Cowardin Wetland Classification System (Cowardin et al. 1979) or the Stewart and Kantrud Wetland Classification System (Stewart et al. 1971) as recommended by Alberta Environments' Provincial Wetland Restoration/Compensation Guide (Alberta Environment 2007). Comparing a restored wetland against the destroyed wetland using the same classification system "helps Alberta Environment determine if a suitable balance is being maintained" (Alberta Environment 2007).
- Create a local DEM using elevation data of reclaimed areas of the mine site. The DEM used for the pre-mine study was created in the early 1980s and an updated, and more accurate DEM would enhance wetland site selection. Wetlands are influenced by groundwater but in the absence of groundwater data, a DEM allows for an approximation of surface hydrology which is useful for hydrologic modeling (White and Fennessy 2005),
- Incorporate existing climate data or climate models predicting long-term forecasts, especially for hydrologic budgets of perched wetlands. Peters et al. (2006) modeled the response of perched wetlands to hydroclimatic conditions using climate data (historical and projected) and concluded that perched wetlands

would evaporate and disappear without floodwater recharge. The disappearance of perched wetlands would be detrimental to the wildlife that are dependent on them for breeding and habitat.

- Model a time-series of potential and future land cover change scenarios to project conflicts of land use with wetlands because land use and land cover change affect wetland function. Naugle et al. (2001) used a landscape approach and GIS to integrate wetland and land cover data with wetland bird habitat models to evaluate and select suitable habitat sites. Results from their study suggest that wetland type and connectivity across regions matter for the waterfowl species that depend on them and preserving that connectivity requires land use planning at a regional scale.
- In addition to aerial photographs, use remote sensing technology, for example Light Detection and Ranging (LIDAR) and Radio Detection and Ranging (RADAR) for future land cover analysis and wetland creation. LIDAR and RADAR can aid with detecting water on the landscape even through forest cover and ground cover. Each image type has benefits and limitations but combined they can simplify wetland mapping efforts and improve the accuracy of a wetland map that can be used for "regulation, input for models, natural resource management, and to quantify wetland function" (Lang and McCarty 2008).

4.5 Literature cited

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