Scaling-up natural resource management in northern landscapes:
Utilizing landscape ecology and remote sensing in Environmental Impact Assessment and wetland monitoring

by

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ABSTRACT

The rapid pace of natural resource development often exceeds our capacity to comprehensively monitor its range of environmental impacts. More recently, increases in both the availability and spatio-temporal scale of environmental geodata have created new opportunities for improving environmental assessment and monitoring. Here I aim to improve upon the conceptual and technical basis for scaling-up assessment and monitoring via exploration of two contrasting landscapes typical of northern Canada. In Chapter 2, I examined different ways of incorporating landscape ecology and ecosystem service approaches into the first phase of Environmental Impact Assessment (EIA) during the scoping process. Using examples from the Skeena River watershed in northwest British Columbia, I first demonstrated how changing the extents of spatial boundaries can potentially misrepresent cumulative impacts. Second, I used network analysis to assess the extent to which seemingly small, localized developments can disrupt landscape connectivity and how historical aerial photographs can help improve restoration planning. Lastly, I present a framework for using regulating ecosystem to better account for water filtration services provided by wetlands. Building upon these insights, in Chapter 3 I utilized a 27-year time series of Landsat imagery to monitor road construction and evaluate subsequent recovery trends in boreal peatlands. I examined long-term trends of wetland-relevant tasseled cap (TC) indices (wetness, greenness, etc.) using a Before/After, Control/Impact study design, which also controlled for the impact of precipitation trends. I demonstrated not only that hydrologic disruption is discernible via remote sensing, but that it persisted for at least 5-years post-disturbance with no signs of hydrologic recovery in several types of wetlands. Furthermore, hydrologic disruptions following road construction were not evenly distributed within wetlands, but rather concentrated near roadside margins. Taken
together, my research provides insight into the possibilities for practical yet rigorous up-scaling of environmental assessment techniques using open-source data. Furthermore, my work also highlights current gaps in implementation of concepts and tools in environmental monitoring, as well as solutions to aid responsible, transparent natural resource development.
LAY SUMMARY

The rapid pace and broad scale of development has often exceeded our ability to monitor and understand the environmental impact. However, expanding availability of geodata has provides new opportunities for environmental assessment and monitoring. My thesis focuses on ways to use different types of spatial data to achieve two different environmental assessment and monitoring outcomes. The first outcome demonstrates how different ideas and tools implemented within Environmental Impact Assessments (EIA) in northern B.C., Canada can create a more reliable and transparent assessment. The second approach utilizes remotely sensed satellite imagery to observe the impacts of road-building on wetlands in remote areas in northern Alberta. My findings show the emerging importance of broad-scale publicly-available data for environmental assessment and highlight opportunities for integrating spatial technology for tracking the impacts of development in sensitive and remote ecosystems.
PREFACE

This dissertation is original, independent work conducted by me in Dr. Sarah Gergel’s Landscape Ecology Lab. I was responsible for the study design and execution with assistance from Dr. Gergel. I conducted the analyses and wrote the manuscript with editing assistance from Sarah Gergel, Dr. John Richardson, Dr. Kevin Hanna and Ira Sutherland.

Chapter 2 [Harker KJ, Gergel, SE, Richardson, JS, Hanna, KS, Sutherland IJ] has been submitted for publication and is currently under review. I was the lead author responsible for original concept formation and manuscript composition. I. Sutherland, S. Gergel, J. Richardson and K. Hanna reviewed and edited the manuscript and assisted with concept formulation.
# TABLE OF CONTENTS

<table>
<thead>
<tr>
<th>Section</th>
<th>Page</th>
</tr>
</thead>
<tbody>
<tr>
<td>ABSTRACT</td>
<td>iii</td>
</tr>
<tr>
<td>LAY SUMMARY</td>
<td>v</td>
</tr>
<tr>
<td>PREFACE</td>
<td>vi</td>
</tr>
<tr>
<td>TABLE OF CONTENTS</td>
<td>vii</td>
</tr>
<tr>
<td>LIST OF TABLES</td>
<td>ix</td>
</tr>
<tr>
<td>LIST OF FIGURES</td>
<td>x</td>
</tr>
<tr>
<td>ACKNOWLEDGEMENTS</td>
<td>xi</td>
</tr>
<tr>
<td>DEDICATION</td>
<td>xii</td>
</tr>
<tr>
<td>I: INTRODUCTION</td>
<td>1</td>
</tr>
<tr>
<td>Remote sensing data can expand the spatio-temporal scale of monitoring</td>
<td>2</td>
</tr>
<tr>
<td>Importance of monitoring northern wetland landscapes</td>
<td>3</td>
</tr>
<tr>
<td>Overview of Research: Two contrasting case studies in northern landscapes</td>
<td>5</td>
</tr>
<tr>
<td>Building on the opportunity to improve wetland monitoring</td>
<td>6</td>
</tr>
<tr>
<td>II: How can lessons from landscape ecology and the use of ecosystem services improve environmental impact assessment scoping?</td>
<td>9</td>
</tr>
<tr>
<td>Introduction</td>
<td>9</td>
</tr>
<tr>
<td>Opportunity 1 - The impact of spatial extent on scoping</td>
<td>14</td>
</tr>
<tr>
<td>Example - Changing the spatial extent changes the number and type of development</td>
<td>15</td>
</tr>
<tr>
<td>Example - Habitat connectivity as a way to determine appropriate spatial boundaries</td>
<td>18</td>
</tr>
<tr>
<td>Opportunity 2 - Learning from the past: shifting baselines and the appropriate temporal scale of an EIA</td>
<td>22</td>
</tr>
<tr>
<td>Example - Historical data can improve the validity of pre-construction baseline assessments</td>
<td>25</td>
</tr>
<tr>
<td>Opportunity 3 - Incorporating regulating ecosystem services: lessons from wetlands</td>
<td>27</td>
</tr>
<tr>
<td>Conclusions</td>
<td>35</td>
</tr>
<tr>
<td>III: Monitoring long-term responses of boreal wetlands to road disturbances using the Landsat archive</td>
<td>37</td>
</tr>
<tr>
<td>INTRODUCTION</td>
<td>37</td>
</tr>
<tr>
<td>Potential impacts of roads on wetlands</td>
<td>38</td>
</tr>
<tr>
<td>Developing a wetland monitoring solution</td>
<td>39</td>
</tr>
<tr>
<td>METHODS</td>
<td>42</td>
</tr>
</tbody>
</table>
LIST OF TABLES

Table 2.1. Benefits and roadblocks to integrating ecosystem services (ES) into environmental impact assessment (EIA) and cumulative effects assessment (CEA). ................................................. 28
Table 2.2: Unpackaging a Regulating ES - Measuring capacity of a wetland for water quality improvement and regulation ............................................................................................................ 33
Table 3.1. Tasseled cap transformation coefficients used for cross TM and ETM/ETM+ sensor consistency........................................................................................................................................ 48
Table 3.2. Impact of roads on TC indices measured in adjacent wetlands at 1-, 3-, and 5-years following construction ........................................................................................................... 59
Table 3.3. Percentage of pixels with structural breaks after road construction detected using BFAST (at a significance level ≤ 5%). ........................................................................................................ 62
Table A-1. Description of the three wetland classification systems used in combination to create the Merged Wetland Inventory (MWI) by Alberta Environment and Parks (AEP). ............ 100
Table A-2. Results of a comparative accuracy assessment between 2012 BAP map and MWI. .................................................................................................................................................. 101
LIST OF FIGURES

Figure 2.1 – Examples of how site boundaries could be modified to exclude or favor certain outcomes in environmental impact assessment (EIA) and cumulative effects assessment (CEA). ........................................................................................................................................................................ 18
Figure 2.2 – Network analysis demonstrates the importance of evaluating impacts to landscape-level connectivity from proposed projects. ........................................................................................................................................................................ 22
Figure 2.3 - Lessons to be learned from historical aerial photographs. ................................................................................................................................. 24
Figure 2.4 – Utilizing historical vegetation mapping in EIA/CEA baselines. ............................................................................................................................... 27
Figure 3.1. Depiction of the hypothesized outcomes of wetland hydrological response to road building disturbance................................................................................................................................. 41
Figure 3.2. An example using CVA to identify road disturbances. ................................................................................................................................. 51
Figure 3.3: Using CVA to determine the year of road construction for 86 sites. ................................................................................................................................. 55
Figure 3.4: Percent change in mean tasseled cap (TC) indices at one, three and five years following road construction relative to control wetlands. ................................................................................................................................. 59
Figure 3.5: Change in TC wetness before and after road building. ................................................................................................................................. 61
Figure 3.6. Temporal distribution of BFAST structural breaks. ................................................................................................................................. 63
Figure 3.7. Spatial distribution of the magnitude of BFAST breakpoints. ................................................................................................................................. 65
Figure 3.8. A violin plot of BFAST breakpoint magnitude in each year following road disturbance. ................................................................................................................................. 66
Figure 3.9. Comparison of the wetland elevation (left, Panel A) to the distribution of positive and negative BFAST breaks (right, Panel B) ................................................................................................................................. 68
Figure 3.10. A violin plot of the up-gradient (left) and down-gradient (right) BFAST breaks. ................................................................................................................................. 69
Figure 3.11 Average monthly precipitation in Fort McMurray from 1985 to 2012 ................................................................................................................................. 76
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DEDICATION

This work is dedicated to my dad, Jim, and my grandpa, Garth. Your presence in my life inspired a lifetime of curiosity and a great appreciation of hard-work.
I: INTRODUCTION

A perplexing and central challenge of ecology is understanding how local impacts to ecosystems contribute to cumulative landscape-level change (Kennedy et al. 2014). Similarly, solving environmental problems across spatial scales, from landscape to regional, is an enduring scientific challenge (Liu & Taylor 2002; Miller et al. 2004; Peters et al. 2006). In ecology, one bridge between local and landscape scales is the discipline of landscape ecology (LE). LE encompasses a diverse range of concepts, tools, and perspectives for assessing broad-scale, as well as long-term, changes in ecosystems. As such, LE themes and approaches have been proposed as effective instruments in “scaling-up” site-based natural resource management (Liu & Taylor 2002).

Traditionally, natural resource management has focused primarily on localized, site-level phenomena (e.g. one oil and gas facility, or a single stand within a managed forest) (Liu & Taylor 2002; O’Higgins et al. 2015). Among the most globally-relevant examples of this site-level focus is the approach taken in Environmental Impact Assessment (EIA). EIA was introduced in the 1970s to inform decision-makers (i.e. funding agencies, governments) and stakeholders of the expected ecological consequences of a particular development (Cashmore 2004; Jay et al. 2007; Morrison-Saunders et al. 2014). By the 1980s, EIA was expanded to include cumulative effects assessment (CEA) as it was recognized that the focus on individual sites was insufficient to account for the combined and interacting effects of different natural resource developments (Duinker and Greig 2006). Despite this widespread recognition of the need to account for multiple stressors occurring over time and space (O’Higgins et al. 2015; Jones 2016), effective implementation is less clear and has lagged behind (Aspinall & Pearson 2000; O’Higgins et al. 2015). Here, one focus of my thesis is to apply different concepts, ideas,
and approaches of landscape ecology to EIA and to improve upon long-term environmental monitoring of ecosystems and landscapes.

**REMOTE SENSING DATA CAN EXPAND THE SPATIO-TEMPORAL SCALE OF MONITORING**

A limiting factor to scaling up local, site-level assessments to a broader spatial extent is the amount and type of data required (Soranno et al. 2015). However, there has been a strong push within the scientific community to make data publicly available, to combine site data into larger databases, to extend monitoring programs, and to standardize data collection efforts for multiple end users. Increasing data availability in recent decades has led to growth in meta-analyses, creation of ecological synthesis centers, use of Geographic Information Systems (GIS) for data storage and analysis, and proliferation of ‘big data’ projects reliant on intensive computer processing (Steiniger & Hay 2009; Soranno et al. 2015).

Among the most important and widely-used open data sources for expanding the spatio-temporal scale of ecological inquiry is remote sensing (RS) imagery (Kennedy et al. 2010; Frohn & Lopez 2017). Remote sensing is used for a multitude of natural resource applications including tracking land-use/land-cover change (Asner & Vitousek 2005), species inventories (i.e. for use in forest harvest planning), and monitoring disturbances ranging from anthropogenic to natural disturbances such forest fire, pine beetle outbreaks, etc. (Gross et al. 2009; Nemani et al. 2009; Kennedy et al. 2009; Kayastha et al. 2012; Thackway et al. 2013). However, a fundamental trade-off exists between image resolution (which impacts the amount of discernable detail) and image processing time and costs (Frohn & Lopez 2017). For example, aerial photography (which ranges by region in both availability and cost) is often acquired at very high spatial resolution yet often requires substantive time for processing and interpretation over vast
areas (Gergel et al. 2007; Morgan et al. 2010). In contrast, lower spatial resolution satellites, such as MODIS, provide broader spatial coverage (at regional to global spatial scales) and twice-daily temporal resolution at no cost, yet are unable to detect fine-scale ecological patterns.

Landsat imagery provides a useful compromise among these trade-offs in ecological interpretation, scale, and cost (Cohen & Goward 2004; Wulder et al. 2012; Wulder et al. 2016). The opening of the Landsat archive for public use by the United States Geological Survey in 2008 (Woodcock et al. 2008) has further revolutionized its utility. The Landsat archive contains over 40 years of observations (since 1972) at a medium resolution (30 m) with repeat coverage at ~16-day intervals for some areas of the world. This expansion of spatial and temporal coverage, provided to the scientific community at no cost, has supported an exponential increase in the scientific use of Landsat (Wulder et al. 2012).

As open data becomes increasingly accessible to scientists at academic institutions, government agencies, and consulting companies, there will be greater opportunity to improve landscape monitoring and bolster landscape-level decision-making in natural resource developments. Some of the greatest opportunities to exploit the utility and benefits of open RS data will likely transpire in historically understudied and inaccessible landscapes, such as those found throughout the vast extent of northern Canada.

**IMPORTANCE OF MONITORING NORTHERN WETLAND LANDSCAPES**

One of the intended achievements of my thesis is to reduce the barriers to landscape-level study of wetlands. Globally, it is estimated that 54 - 87% of the world’s natural wetlands have been lost since the start of the 18th century (Davidson 2014). Wetlands provide many different ecosystem services (ES) including regulating water quality (i.e. nutrient uptake, biogeochemical processing, and sediment retention) and water quantity (i.e. groundwater recharge, maintenance
of hydrological connectivity, and runoff and floodwater retention) across a diversity of landscapes (Zedler 2003; Vanderhoof et al. 2016). North temperate and boreal wetland ES are globally important as their dynamics impact global carbon budgets and provide large, shallow surface-water storage (Morris et al. 2011; Waddington et al. 2015). Much of Canada’s vast northern boreal region is dominated by wetlands, which represent nearly 20% (96,500 km²) of the world’s total wetland area (Webster et al. 2015). Industrial activity in the northern boreal, such as forestry, energy development, as well as mining, is likely affecting wetland condition and delivery of important ES (McLaughlin & Cohen 2013; Webster et al. 2015). However, wetland monitoring remains difficult due to seasonally dynamic hydrologic conditions and the great areal extent and remote location of northern wetlands (Toyra & Pietroniro 2005). As such, assessing and predicting the impact of development projects through EIA processes can be fraught with difficulties and often such assessments lack a clear and objective way forward.

Fortunately, both Landsat imagery and historical aerial photographs can provide insight into long-term dynamics of wetland landscapes and aid EIA especially in data-limited northern regions. Research on the use of RS for wetland mapping and monitoring is extensive (see reviews in Ozesmi & Bauer 2002; Adam et al. 2010; Tiner et al. 2015) as the temporal frequency, wide spatial coverage, and low cost of Landsat satellite imagery (for example) is ideal for monitoring the broad scale effects of natural resource development.

Aerial photography contains the longest spatially contiguous historic imagery of earth’s surface, dating back to the 1930s (or earlier) in most places (Morgan et al. 2010). As such, it has been used for decades to characterize wetland extent, vegetation type, inundation status, and to guide wetland classification (Frayer et al. 1983; Hardisky & Klemas 1983; Tiner et al. 2015). Aerial photo databases provide an opportunity to fill in data gaps and determine previous
vegetative/wetland conditions and trends in data-limited areas. As such, aerial photos and satellite imagery have been used in both Canada and the United States, to create the Canadian Wetland Inventory (Ducks Unlimited 2018) and the National Wetlands Inventory, respectively (Dahl and Watmough 2007).

When used in tandem, the combination of aerial photography and Landsat provide an unparalleled opportunity to assess landscapes at high spatial resolution, over longer time frames, and evaluate the impacts of activities such as land cover change. One of the goals of this thesis is to better integrate historical photography and long-term satellite archives to improve our assessment of landscape connectivity, ecosystem services and hydrologic dynamics in northern areas.

OVERVIEW OF RESEARCH: TWO CONTRASTING CASE STUDIES IN NORTHERN LANDSCAPES

EIA processes and subsequent outcomes are a critical component of natural resource development and decision making. Thus, in Chapter 2, I explore opportunities to improve the rigor and utility of EIA by incorporating principles and approaches of landscape ecology and ecosystem services. With a focus on the Canadian EIA scoping process, I explore hypothetical, yet realistic, site- and landscape-level situations to demonstrate how lack of integration of scientific principles can impact EIA outcomes using the backdrop of the Skeena River watershed in northern BC.

The Skeena River watershed is a mountainous, primarily forested region and is the only major un-dammed river system in the northern hemisphere (Nilsson et al. 2005; Richardson & Milner 2010). Anthropogenic disturbances are relatively recent and of much smaller extent compared to similar watersheds in southern B.C. (Lewis 2008; Jones et al. 2010). While timber
extraction largely began in the 1970s, currently the Skeena-Stikine region (which comprises the majority of land area in northwestern B.C.) is facing the highest number of proposed major natural resource development projects in the province (OAG BC 2015). Project proposals include new mines, liquefied natural gas (LNG) facilities, crude oil facilities (including offshore oil development) and various hydroelectric projects (OAG BC 2015). This growing development pressure on the highly biodiverse and productive ecosystems in the Skeena create an interesting and timely study site in consideration of the scale of concepts and tools used in natural resource decision-making.

In Chapter 2, I first demonstrate how changing the spatial extent of EIA boundaries can potentially misrepresent cumulative impacts through inappropriate exclusion of surrounding natural resource development projects. Second, I use a network analysis approach to show how even a seemingly small, localized development area can disrupt overall landscape habitat connectivity. I also reveal how tools for characterizing riparian baseline vegetation (using historical aerial photos), can potentially better convey the effects of past stressors and help to improve the efficacy of restoration planning. Finally, I show how the lens of regulating ecosystem services can better account for water filtration services provided by wetlands in an EIA context. These approaches are straightforward to implement using publicly available data, thus provide sensible opportunities to improve EIA and concomitant decision-making. As public involvement in and skepticism of EIA grows, the need for objective, transparent, and rigorous biophysical assessments is both relevant and timely.

BUILDING ON THE OPPORTUNITY TO IMPROVE WETLAND MONITORING

In Chapter 3, I investigate the landscape and site-level effects of building roads through wetlands in northern Alberta, Canada using a Landsat imagery time series. I evaluate a suite of
vegetation indices designed to capture long-term hydrological changes using a paired approach comparing road-impacted and non-impacted wetlands. The geology, climate, and history of natural resource development in Alberta’s Central Mixed Natural Subregion (CMNS) make it an ideal location to study the impact of road building on wetlands. Deep glacial sediment deposits, gradual relief, and cooler temperatures result in a tremendous capacity for water storage and distribution via its large complex of wetlands (Devito et al. 2012; Devito et al. 2005). Similar to the Skeena River watershed, much of this area remained largely undisturbed until relatively recently due to its remote location and inaccessibility. However, since the inception of natural resource extraction in the 1960s (particularly petroleum), the scale of development has expanded considerably (Boutin & Carpenter 2017). To appreciate the scale of such anthropogenic impacts, the area of boreal peatlands in northern Alberta impacted solely by open-pit bitumen mining approached 30,000 ha (Rooney et al. 2011).

My exploration of two case-studies (Skeena River in B.C. and boreal peatlands of Alberta) provides much needed insight into the challenges and opportunities for landscape-level analysis of natural resource development in remote areas where biophysical information remains limited. Because the footprint of disturbance on the northern Alberta landscape far exceeds the current footprint of natural resource development in the northern B.C. Skeena case study, these case study locations span a compelling range of transitioning northern landscapes in different phases and with differing types of impacts. Through the focus on wetlands and riparian areas (in parts of Chapter 2 and in Chapter 3), I tackle the recurring challenge of creating objective and repeatable ecosystem monitoring approaches for complicated hydrological impacts of natural resource development. In recognition of these complexities and in order to encourage wider adoption of these geospatial practices, all of the data in Chapter 2 and 3 are from publicly
available sources. Furthermore, through an emphasis on FOSS (free and open source software) - compatible approaches, I aim to demonstrate the ease and efficacy of the proposed approaches for a broad, non-academic audience.
II: How can lessons from landscape ecology and the use of ecosystem services improve environmental impact assessment scoping?

INTRODUCTION

The impacts of rapid natural resource development are cumulative and widespread, yet many management decisions, such as environmental impact assessment (EIA), are made at the project level. As a key, universally recognized instrument in environmental management (Morgan 2012), EIA is a critical place to consider the landscape perspective. The EIA process has been a consistent, if somewhat contentious, part of natural resource development in countries around the world since its inception in 1970 in the United States (Cashmore 2004; Jay et al. 2007; Morrison-Saunders et al. 2014). Project-based environmental impact assessments (EIA) were originally proposed as a way of informing decision-makers and stakeholders of expected and/or potential ecological consequences of development (Cashmore 2004; Jay et al. 2007). EIA, in some form, is now practiced in all member countries of the United Nations, with two exceptions, the Democratic People's Republic of Korea and South Sudan (Morgan 2012; Jones 2016). By the 1980s, the need for cumulative effects assessment (CEA) was recognized as EIA assessed the impacts of projects in isolation and did not account for the combined effects of human activities (Duinker and Greig 2006). The introduction of CEA, as a sub-field within EIA, expanded the EIA scope to account for the effects of multiple stressors that occur over time and in space (Jones 2016). The growth of EIA (including CEA) worldwide and the level of engagement amongst researchers, practitioners and public stakeholders reinforces the importance of the process and the need for continuous updates to practice.
Perceptions of the role and effectiveness of EIA and CEA (referred to collectively as EIA) varies amongst different actors (Morrison-Saunders and Bailey 2003; Morrison-Saunders and Sadler 2010; Runhaar et al. 2013). One persistent area of contention is the level of integration and quality of science (particularly ecology) used in EIA practice (Morrison-Saunders and Bailey 2003; Morrison-Saunders and Sadler 2010). On one hand, there have been great improvements in the incorporation of ecological science into EIA in the decades since inception (Greig and Duinker 2011) and the role of disciplines such as environmental toxicology have been fundamental in creating technical regulations and guidelines (i.e. drinking water contaminant levels, air quality regulations) used in EIA. However, in other areas the incorporation of ecological science has lagged behind and thus several opportunities for improved EIA practice are apparent. I (and many others) advocate that strong science in EIA is imperative to greater transparency, informed decision-making, and maintenance of ecosystem functioning.

The differences in approach and implementation between ecology and EIA practice has led to several disconnects. First, there are differences in the scale considered, both temporal and spatial, in EIA compared to ecological research of environmental impacts (Raudsepp-Hearne and Peterson 2016). Project-based and regulatory EIA typically focuses on only one location over a relatively short, near-term contemporary timeframe whereas the scale of research is closely linked to the phenomenon of interest. Second, EIA was developed as a regulatory and political mechanism without a scientific framework in place and it has subsequently evolved in response to practice and regulatory requirements instead of scientific advances (Cashmore 2004; Jones 2016). Third, EIA processes may face limited resources in terms of time (due to client deadlines), money (particularly for smaller, single-site projects) and data (especially in remote
areas) to complete comprehensive assessments. Fourth, despite the potential guidance provided by peer-reviewed science (both data and methods) to maintain the objectivity of EIA (Greig and Duinker 2011), in practice such guidance is often not utilized. The sheer magnitude of published literature, the diversity of methodologies, and constantly evolving nature of scientific (and other) research has made adoption of the most recent science into EIA difficult (Jones 2016). The lack of implementation of ecology into EIA reviews often means that best management practices and judgement of long-time practitioners are used to determine the effects of a project as well as the magnitude of those effects (Greig and Duinker 2011; Noble 2015). Despite their value, reliance on practitioner judgement may introduce personal bias and over-emphasize the relevance of past approvals with less applicability to current projects (Iftekhar and Pannell 2015).

One fundamental stage of EIA that demonstrates the disconnect between ecology and EIA is the scoping process (Snell and Cowell 2006). Scoping determines which ecological effects/impacts are important to assess and frames the spatio-temporal boundaries of an assessment based on expected or predicted environmental impacts. The scoping phase provides a preliminary understanding of how impacts will interact with nearby disturbances (Noble 2015). The scoping process also typically identifies alternative project locations and designs. Despite its fundamental importance to EIA effectiveness, scoping has been under-researched and is often overlooked (Snell and Coward 2006). Scoping of an EIA presents the difficult task of balancing precaution (i.e. including every possible environmental impact, cumulative impacts and interactions) and project efficiency (i.e. leaving out important impacts due to fiscal and/or time constraints) (Mandelik et al. 2005; Snell and Coward 2006). These competing pressures combined with the absence of a scientific framework to ensure accountability, unclear linkages
to scientific practice, and an established reliance on long-time practitioners, have led to disconnects between ecology and scoping practice.

The EIA scoping stage is practiced globally but the methods vary. For example, in British Columbia (BC), Canada (as well as other provinces in Canada), scoping is a process where some parameters are defined and required in regulations and others are decided by proponents and through negotiation with stakeholders and agencies (BC Environmental Assessment Office [EAO] 2013; Hanna 2016). This occurs through the selection of Valued Ecosystem Components (VECs), defined as components of the environment that are considered to have important scientific, environmental, economic, social and/or cultural importance to project stakeholders (BC EAO 2013; Noble 2015; Hanna 2016). This differs from scoping in other jurisdictions (e.g. the USA and UK) where it typically defines a list of potential issues, instead of selecting important ecosystem components (Mulvihill 2003; Snell and Coward 2006). Regardless of method, the impacts or ecosystem components selected for assessment not only dictate the outcome, but also the type and extent of mitigation measures eventually proposed and implemented (Mandelik et al. 2005; Snell and Cowell 2006).

Recognizing both the importance and limitations of EIA, I propose that an effective way to address some areas of divide between ecology and EIA scoping is through the lens of landscape ecology (LE) and ecosystem services (ES). LE and ES approaches invoke useful comparison to the disconnects discussed above in EIA for several reasons. LE is an interdisciplinary field that aims to study the impacts of anthropogenic disturbance on landscape structure, function, and change and centered on the importance of spatial and temporal scale (Liu and Taylor 2002; Turner and Gardner 2015). The ES approach also assesses the consequences of anthropogenic ecosystem change and provides a scientific framework to identify, understand,
and monitor how complex attributes of ecosystems provide benefits to people (MA 2005; De Groot et al. 2010; Sutherland et al. 2017). Collectively, both LE and ES utilize robust and easily accessible methods and tools with a strong foundation in real-world application (Liu and Taylor 2002; Pearson and McAlpine 2010). Additional benefits to incorporating LE and ES into EIA include quantifying uncertainty for decision makers, the established use of Geographic Information Systems (GIS) to synthesize information and communicate effectively with a wider audience, and methods that are widely scalable to match project needs and fit into regional, national and global analysis (Liu and Taylor 2002; Pearson and McAlpine 2010). Focus on ES and LE as analogous disciplines to EIA can also streamline information and adoption of scientific advancements into practice.

To demonstrate the usefulness of incorporating LE/ES into EIA, I adapt well-established concepts and tools into the EIA scoping stage. Opportunities where LE and ES can be readily integrated into EIA scoping are highlighted as examples. Each example utilizes a different tool or framework to analyze the hypothetical effects of a site-based, natural resource development in BC, Canada. The first opportunity for integration is selecting the spatial scale (or extent) of the EIA, the second considers the appropriate temporal scale, and the third explores the benefits of using a regulating ES framework. To emphasize the accessibility and ease of incorporating these approaches into EIA, which is often resource-limited, I use publicly available data and free and open software (FOSS), with the exception of ArcGIS, which can be substituted for FOSS alternatives. Each opportunity for integration is presented by introducing the LE/ES concept, discussing how it enhances the rigour of the EIA process and then providing example(s) of how the LE concept could be better integrated.
Our examples are set in BC because the EIA approach taken in BC and in Canada has global ecological implications. Canada’s land base covers a massive 9.985 million km² extent, which includes 5.4% of the world’s renewable surface water supply, and is rich in natural resources (Webster et al. 2015). As such, Canada’s economy is largely based on natural resource extraction (representing 12.5% of the GDP in 2009) and the spatial footprint of natural resource development is expected to increase in northern boreal areas by 50-60% by 2030 (Price et al. 2013; Webster et al. 2015). Despite the expected increase in development, the boreal zone of Canada (~3.09 million km²) is currently one of the world’s largest remaining forest with minimal anthropogenic disturbance (Andrew et al. 2013; Price et al. 2013; Webster et al. 2015). Therefore, along with opportunity for economic development, there is also great opportunity to make important changes in assessment processes to ensure that when development occurs I maintain landscape-level ecosystem functioning and limit negative global impacts.

**OPPORTUNITY 1 - THE IMPACT OF SPATIAL EXTENT ON SCOPING**

The impact of spatial extent in influencing measured ecological processes is a fundamental concept in landscape ecology and in all natural sciences (Turner and Gardner 2015). Failure to consider the appropriate spatial scale of ecological processes has led to many management failures (i.e. transboundary pollution, vulnerability to flooding events, climate change, population-level fisheries collapses) (Cash et al. 2006). Local, project-based decision-making, such as EIA approval, is especially susceptible to mismanagement as a result of scale decisions. Contributions to long-term changes such as climate change are often considered beyond the scope of individual project assessments (Gibson et al. 2016). However, the accumulation of and interaction among stressors from nearby and subsequent developments can
create larger-scale environmental problems, requiring more rigorous, holistic, and comprehensive approaches to CEA.

EIA outcomes (particularly for CEA) are inherently shaped by the spatial extent over which they are assessed (João 2002; Karsten et al. 2007; Raudsepp-Hearne and Peterson 2016). The spatial extent considered at the scoping stage is a complex, context dependent and critical decision. For these reasons, combined with the general paucity of data available at this stage, the required spatial extent of EIA and CEA is poorly, if at all, defined by legislation in Canada. Decisions about spatial extent in Canada can be somewhat arbitrary and often defined by the project proponent (Lebel 2006). To demonstrate the implications that spatial extent has on project outcomes, I provide two examples that show the effects of modifying the spatial boundary of CEA and use landscape connectivity to gain a greatly enhanced spatial perspective.

**Example - Changing the spatial extent changes the number and type of development**

A change in spatial extent of a project study area affects the number and type of natural resource development projects located in proximity to a newly proposed project. Because CEA aims to incorporate impacts of existing developments (and those reasonably foreseeable) within a general vicinity, landscape context becomes increasingly relevant (Therivel and Ross 2007). A simple exercise in expanding the site boundaries of a project demonstrates that the cumulative effects of a given project and the relevant ecological components assessed potentially change from few to many with changes in spatial extent (Figure 2.1). While there are efforts to address the impact of spatial extent choice through CEA, such efforts have been limited because CEA is a requirement for project-based EIA and is not well connected to other projects in close proximity and to larger temporal and spatial landscape change (Noble 2015). Furthermore, a lack of regulatory oversight on spatial extent at this stage may allow for the boundaries of the study...
area to be modified to better suit particular stakeholders (Lebel 2006; Karsten et al. 2007).

Although simply increasing the spatial extent of analysis (as shown for demonstrative purposes only in Figure 2.1) is not meaningful in practice, it is used to show the importance of this decision for CEA and for transparency with stakeholders.
Figure 2.1 – Examples of how site boundaries could be modified to exclude or favor certain outcomes in environmental impact assessment (EIA) and cumulative effects assessment (CEA). Results show ensuing differences in the number and type of development detected at each site. For example, a CEA at Site 3 would discern only two forest cut blocks at a 2.5 km radius compared to five active forest cut blocks and 19 contaminated sites at a 5 km radius, thus drastically changing the environmental effects/impacts included in the assessment. Although simply expanding spatial extent is not recommended, it is shown here to provide insight into the importance of this decision in the scoping process. Figure based on three randomly selected EIA reviews in the pre-application stage (Data BC website) as of June 2017.

In order to provide transparent and accurate information, the rationale and consequence of choosing a particular spatial extent must be made explicit. I argue that the spatial extent chosen for EIA and CEA should go beyond the demonstrated example in Fig 2.1 and ultimately be a function of the spatial extent over which affected ecological processes operate (i.e. water quality is assessed at the spatial extent over which contaminants historically and potentially travel), the proximity of prior, existing, and future projects, and this decision should be driven by data. While a requirement to assess a more relevant spatial extent to capture ecosystem processes could be perceived as an additional roadblock for EIA efficiency, it can also create opportunities for landscape-level mitigation, such as restoring critical habitat patches lost from past development. While there is no universally correct approach to determining an appropriate spatial extent, explicit consideration of the impacted ecological processes can help select a defensible answer to this issue.

**Example - Habitat connectivity as a way to determine appropriate spatial boundaries**

Habitat connectivity maps are one approach to providing greater spatial perspective. Connectivity refers to the spatial arrangement of a land cover (or habitat) type and reflects whether movement among patches is facilitated or impeded (Belisle 2005; Turner and Gardner 2015). Connectivity is essential for gene flow and dispersal of individuals and thus fundamentally underpins conservation biology and landscape ecology (Coulon et al. 2004). One
approach for measuring connectivity is network analysis. Network analysis denotes a series of “nodes” or habitat patches potentially connected by links (Urban and Keitt 2001; Kupfer 2012). When two nodes are connected through a link, an exchange of energy or material flow is assumed (Urban and Keitt 2001) with important implications for natural resource management. Conducting a habitat connectivity analysis during the EIA application process can quantify potential effects of even a small project on overall regional connectivity (Figure 2.2). Such an analysis can help to quantify the consequences of choosing a particular spatial extent, provide a better understanding of the effects of disturbance on a regional landscape scale, and help weigh options for alternative project locations.
Fig 2 Panel A
Figure 2.2 – Network analysis demonstrates the importance of evaluating impacts to landscape-level connectivity from proposed projects.

Panel A - Conceptual network connectivity diagram before and after disturbance. Looking exclusively within the site boundaries, the project may be perceived as having little impact on the overall habitat area (left). However, when viewed within the context of overall landscape connectivity, removal of that specific habitat node disconnects two large areas of previously connected habitat and greatly reduces overall connectivity (right). Modified from Urban and Keitt 2001.

Panel B – Mapped results of network connectivity analysis of critical habitat patches for marbled murrelet (Brachyramphus marmoratus), a threatened coastal bird species in the Lakelse Watershed near Terrace, BC (Falxa & Raphael 2016). Habitat patch size and proximity was evaluated using CONEFOR connectivity analysis software (Saura & Torné 2009) assuming an average dispersal distance of 100 m. The results of CONEFOR analysis showed which habitat patches are the most important in terms of overall connectivity of the landscape (purple) and which habitat patches are the most important as connectors from patch to patch (orange). If a project is located within one of the important habitat patches, there is justification for inclusion as a VEC and/or it can prioritize alternative locations for the project.

OPPORTUNITY 2 - LEARNING FROM THE PAST: SHIFTING BASELINES AND THE APPROPRIATE TEMPORAL SCALE OF AN EIA

Historical ecology provides insight into long-term patterns and processes in ecosystem disturbance regimes, response and recovery following disturbance (Swetnam et al. 1999; Higgs et al. 2014) and more recently, analysis of ecosystem services (Renard et al. 2015; Sutherland et al. 2016; Tomscha et al. 2016). Among the most prevalent sources of historical ecological data is aerial photography, typically providing coverage since the 1930s or certainly 1950s in most regions (Morgan et al. 2010). Additional sources include remote sensing imagery (i.e. Landsat archive) (Kennedy et al. 2014), field assessment reports (i.e. soil surveys, maps), long-term ecological research projects (i.e. monitoring data, data loggers, weather records) sediment and ice core analysis, tree ring data, and local ecological knowledge (LEK) accounts (Swetnam et al. 1999; Tomscha and Gergel 2015; Tomscha et al. 2016). Understanding the history of a landscape is important for characterizing the effects of land-use and land cover change, response to human and natural disturbance, and to determine if time lags in ecosystem responses occur (Burgi et al.
Determination of ecosystem ‘baselines’ is one of the fundamental purposes of the scoping process (Hanna 2016). The goal is to determine baseline conditions of ecological components prior to disturbance and to predict the adverse environmental effects relative to this baseline (BC EAO 2013). EIA baselines are determined by on-site fieldwork and publicly available data (i.e. weather monitoring stations, information from projects nearby, or other assessments) often collected over a short time span. Importantly, the idea of pre-construction baseline conditions used in EIA differs greatly from the idea of baselines used in ecology. A key insight of historical ecology is the idea of shifting baselines and shifting baseline syndrome (Pauly 1995). As each new generation interacts with an ecosystem, their perceived baseline is different and often more degraded than the generation before. This potentially results in the use of inappropriate reference points for assessing environmental change (Pauly 1995). Applying this to an EIA baseline would better support careful consideration of the temporal boundary for the assessment and also introduces potential for project proponents to reduce cumulative effects by restoring previously degraded ecosystems as part of project mitigation, restoration or decommissioning.

I recommend that to develop a greater understanding of the ‘baseline’ in a natural system, the temporal scale considered in EIA should include information about the past natural and anthropogenic ecosystem pressures. Analysis of historical landscapes can also help reveal the often unexpected and complex effects of decisions and past policy in shaping landscape condition (as seen in Figure 2.3) (Higgs et al. 2014; Burgi et al. 2015; Renard et al. 2015). One method of incorporating past landscape pressures is through the use of historical aerial photos.
Figure 2.3 - Lessons to be learned from historical aerial photographs.

The photos demonstrate how considering historical baselines can prevent environmental decision-makers from making similar mistakes in a contemporary setting. The photo comparison shows how a forestry service road was built along this section of the Lakelse River, a tributary of the Skeena River near Terrace, BC, after 1937 and was built without proper cross drainage implements. As such, the road caused water to pond and created a large wetland feature in this area. This potentially impacted natural flow of runoff to the river, affected nutrient retention and release timing, and other biophysical processes.
Example - Historical data can improve the validity of pre-construction baseline assessments

Incorporating historical aerial photos and vegetation information (Figure 2.4) helps reconstruct a more realistic baseline, particularly in sensitive ecosystems. Development of a more accurate baseline is essential to characterize the cumulative effects that have previously occurred on a landscape and provide valuable insight into how the ecosystem has responded to past disturbances. In recognition of the processing and interpretation time required for aerial photo analysis, targeted analysis of important and/or sensitive ecological areas, such as riparian zones (Figure 2.3 and 2.4), can provide an important starting point for incorporating historical information. Although this type of analysis may not be relevant to broad spatial analyses, targeted historical analysis is essential for learning from past mistakes and ensuring accountability in initial project design and for restoration methods and timelines. Such information will increase the validity of proposed mitigation and provide more realistic insight for management decisions.
Figure 2.4 – Utilizing historical vegetation mapping in EIA/CEA baselines.

A 10 km stretch of the Skeena River in northwestern British Columbia. Vegetation species composition is depicted on the map as classified (using Terrestrial Ecosystem Mapping protocols [BC Ministry of Environment 2000]) from aerial photographs at two different times. Figure 2.4 shows altered river channels and conversion of a primarily coniferous forest in 1947, to early-successional deciduous forests. Information derived from aerial photograph chronosequences such as this provides information about forest recovery patterns, changes in ecosystem services such as carbon storage, the frequency of natural disturbance events, and help to construct proper baseline condition for EIA and CEA. Additional time steps should be used to target specific disturbances and monitor recovery timelines. Data provided through BC Open Data; TEM polygon identification completed by de Groot et al. 2005.

OPPORTUNITY 3 - INCORPORATING REGULATING ECOSYSTEM SERVICES:

LESSONS FROM WETLANDS

One discernable and widespread limitation of EIA in BC (and in other areas) is the exclusion of the ecosystem services (ES) concept. First introduced in the 1970s (Millennium Ecosystem Assessment [MA] 2005), ES refer to the benefits received by humans from ecosystems and are often classified into three categories: provisioning, cultural, and regulating services (MA 2005; Fisher et al. 2009; Haines-Young and Potschin 2013). Conceptually, the scoping stage - for example the BC case of using VECs - is similar to the idea of ecosystem services in that both terms refer to aspects of ecosystems with inherent importance. Both terms elicit a need for conservation and/or management. Despite the staggering rise of ES concepts in scientific research (Fisher et al. 2009), practical application of ES into EIA has lagged for several reasons (Table 2.1) (de Groot et al. 2010; Seppelt et al. 2011; Baker et al. 2013). However, the benefits of the ES approach (Table 2.1) for use in EIA are becoming increasingly recognized and operationalized internationally (World Resources Institute [Landsberg et al. 2011; Landsberg et al. 2013]) and in Canadian environmental policies and initiatives (i.e. Alberta Biodiversity Monitoring Institute [Habib et al. 2017]; Ecosystem Services Toolkit for Federal, Provincial and Territorial Governments [Preston and Raudsepp-Hearne 2017]).
Table 2.1. Benefits and roadblocks to integrating ecosystem services (ES) into environmental impact assessment (EIA) and cumulative effects assessment (CEA).

<table>
<thead>
<tr>
<th>Benefits to ES incorporation into EIA/CEA</th>
<th>Source</th>
</tr>
</thead>
<tbody>
<tr>
<td>Systems approach</td>
<td>Baker et al. 2013, Munns et al. 2015, Sutherland et al. 2017; Landsberg et al. 2013</td>
</tr>
<tr>
<td>ES frameworks extend beyond traditionally measured discrete indicators and towards a more holistic or connected approach. This includes delayed and indirect benefits experienced from regulating ES.</td>
<td></td>
</tr>
<tr>
<td>Stakeholder engagement</td>
<td>Baker et al. 2013; Munns et al. 2015; Landsberg et al. 2013</td>
</tr>
<tr>
<td>The ES approach presents a more inclusive/effective platform for public engagement. The systems approach more closely reflects the values and spiritual beliefs of many First Nations communities.</td>
<td></td>
</tr>
<tr>
<td>Foundation in science</td>
<td>MA 2005; Seppelt et al. 2011</td>
</tr>
<tr>
<td>ES research and peer-reviewed scientific literature is wide-spanning and continues to grow rapidly. Over 1300 scientists contributed to the MA project.</td>
<td></td>
</tr>
<tr>
<td>Rise of ES mapping literature</td>
<td>Martinez-Harms &amp; Balvanera 2012</td>
</tr>
<tr>
<td>ES maps are useful tools for decision makers and communicating science with the public.</td>
<td></td>
</tr>
<tr>
<td>Transparency in decision-making</td>
<td>Munns et al. 2015</td>
</tr>
<tr>
<td>Framing issues as explicit benefits and/or loss of benefits to humans promotes greater transparency.</td>
<td></td>
</tr>
<tr>
<td>Linkages to economic analysis of ES are becoming more established and defensible. Explicit valuation is advantageous for decision makers when discussing economic trade-offs of projects.</td>
<td></td>
</tr>
<tr>
<td>Complexity &amp; Lack of Guidance</td>
<td>Baker et al. 2013</td>
</tr>
<tr>
<td>Implementation is complicated with many complex and interacting drivers. Complexity creates uncertainty in analysis, which is viewed negatively outside of the scientific community.</td>
<td></td>
</tr>
<tr>
<td>Unfamiliarity</td>
<td>Baker et al. 2013</td>
</tr>
<tr>
<td>ES language and frameworks continue to evolve and may not be familiar to managers and policy makers working on EIA.</td>
<td></td>
</tr>
<tr>
<td>Difficult to Quantify</td>
<td>de Groot et al. 2010</td>
</tr>
<tr>
<td>Lack of a consistent, universal set of indicators and methods to quantify different types of ES.</td>
<td></td>
</tr>
<tr>
<td>Resource Intensive</td>
<td>Baker et al. 2013</td>
</tr>
<tr>
<td>Comprehensive analysis can be very resource intensive.</td>
<td></td>
</tr>
<tr>
<td>Inadequate Risk Assessment</td>
<td>ES technically require human benefit, which may lead to exclusion of important small-scale processes and cause inadequate assessment of risk. I further discuss in the ES section that this problem could be overcome by analysis of regulating ES.</td>
</tr>
<tr>
<td>---------------------------</td>
<td>------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------</td>
</tr>
<tr>
<td>Resistance to Change &amp; Legality</td>
<td>Environmental managers and practitioners are resistant to changes in process due to the additional time and money required to assess ES and due to widespread aversion to legal challenges.</td>
</tr>
</tbody>
</table>
The idea of using ES in EIA is not new and was proposed as early as 1992 (de Groot 1992; Slootweg et al. 2001). In support of this, several studies/organizations have effectively outlined specific guidelines and steps for selecting and integrating ES into EIA (Slootweg et al. 2006; Landsberg et al. 2011; Brownlie et al. 2013; World Resources Institute 2013; Preston and Raudsepp-Hearne 2017). Building upon this foundation of work, I posit one particularly advantageous way to integrate biophysical ES concepts into EIA practice in BC and Canada is through a regulating ES framework. Regulating ES characterize the underlying structure and feedback mechanisms of ecosystems that maintain a safe and desirable range of environmental conditions (Villamagna et al. 2013; Sutherland et al. 2017). Regulating ES act to mediate flows of energy and matter (e.g., flooding, soil erosion, greenhouse gases), moderate the release of harmful substances (e.g., waste, contaminants), and control population dynamics (e.g., pest control, disease outbreaks) (Haines-Young and Potschin 2013). Regulating ES are typically left out of assessments because they are perceived as challenging to conceptualize, seem difficult to measure (Guerra et al. 2014) and they often represent long-term processes (e.g. soil erosion prevention, climate change and regulation, flood regulation, etc.) which can be overlooked in project-by-project review processes. However, regulating ES interact together at a landscape scale to help buffer downstream impacts from projects. For example, declines in water quality or fish productivity could be tempered through maintenance of riparian forest (Gunderson et al. 2010). Failure to characterize the impacts on regulating ES in EIA, and especially in CEA, can lead to delayed impacts (i.e. climate change) to human well-being and decreases the ability of regulating ES to act against future pressures (Sutherland et al. 2017).

Regulating ES frameworks quantify and monitor a suite of interacting indicators, often at different spatio-temporal scales (Sutherland et al. 2017). For example, potential for soil erosion
could be measured by monitoring and mapping areas of soil loss, rainfall, soil erodibility, vegetation cover, grazing practices, logging pressures and other contextual indicators (Guerra et al., 2014). Failure to characterize impacts on regulating ES, and especially in CEA, can lead to delayed impacts (i.e. climate change) to human well-being and decreases the ability of regulating ES to act against indirect effects, cumulative impacts, and future pressures (Sutherland et al. 2017). Their inclusion, however, would lead to more robust appraisal of potential adverse and residual effects of natural resource developments. An example from wetlands science is provided next.

**Wetlands and water filtration**

Maintenance of water quality by wetlands is a regulating ES that is typically not explicitly quantified in an EIA. Wetlands act as a retention and removal zone for contaminated and/or turbid runoff resulting from livestock, human waste, fertilizer use, and forest harvest practices (Cohen et al. 2015). Wetland water quality regulation processes include retention of nutrients, organic and inorganic carbon and sediments and removal of reactive nitrogen (through nitrification, denitrification and mineralization) and phosphorus (through mineralization) (Zedler 2003; Knox et al. 2008; Bernal & Mitsch 2013; Wolf et al. 2013; Cohen et al. 2015). The loss of water quality regulation by wetlands, especially when located near other inland water bodies, has the potential to lead to algal blooms and eutrophication and seasonal hypoxia (low oxygen) in areas with pre-existing high nitrate and phosphorus loads (Zedler 2003; Brander et al. 2011).

In most Canadian jurisdictions, however, wetlands are included in EIA only if direct draining, infilling, or temporary disturbances (e.g. linear pipeline right-of-ways) are expected. Indirect effects (such as increased sedimentation, changes in hydrologic regime et al.) are not considered (Noble et al. 2011). In an EIA context, wetlands are included as important ecological
components in that they provide habitat for birds and amphibians, spawning grounds for fish, and presence of rare vegetation. Unfortunately, wetland regulating ES such as carbon sequestration, flood control or water quality improvement are largely dismissed as falling beyond the scope of an individual project, due to their complexity and the temporal and spatial scale at which the effects are observed (i.e. carbon sequestration is measured locally and valued globally) (Brander et al. 2013).

Wetland removal, when assessed independently or combined with other anthropogenic disturbances, can have a range of effects on surrounding water quality that can be difficult to precisely predict (Trebitz et al. 2007). Despite this, I propose that a regulating service framework can be incorporated in EIA in an efficient way using reasonable indicators (Table 2.2). The number of indicators needed to characterize water quality regulating services for any given wetland will be different and context dependent. However, the breakdown of a regulating service into measurable indicators (such as in Table 2.2) provides tangible insight into the maintenance or loss of ecosystem benefits that may result from wetland removal or disturbance at a landscape scale.
<table>
<thead>
<tr>
<th>Type of Process</th>
<th>Indicator</th>
<th>Importance</th>
<th>Quantification</th>
<th>Method/Data Source</th>
<th>Reference</th>
</tr>
</thead>
<tbody>
<tr>
<td>Hydrological</td>
<td>Catchment area</td>
<td>Determines amount of run-off capacity entering wetland system</td>
<td>Area of catchment, runoff capacity</td>
<td>GIS area calculations, hydrometric data</td>
<td>Cohen et al. 2015</td>
</tr>
<tr>
<td></td>
<td>Timing, frequency and severity of flood-events</td>
<td>Reduces retention time of contaminated run-off, which reduces ability to retain and remove contaminants of concern.</td>
<td>Peak value, event duration and timing, probability of extreme events</td>
<td>Regional/county information</td>
<td>Wolfe et al. 2013; Stewardson et al. 2003</td>
</tr>
<tr>
<td>Biogeochemical</td>
<td>Dissolved Organic Carbon (DOC)</td>
<td>Acts as a substrate for microbial activity (for denitrification and P assimilation), binds nutrients and metal sequestration</td>
<td>DOC concentration, reactivity or lability (several different methods), absorbance (color)</td>
<td>Lab analysis of water samples</td>
<td>Stewardson et al. 2003; Pinney et al. 2000</td>
</tr>
<tr>
<td></td>
<td>Iron (Fe)</td>
<td>Enhances complexation of contaminants (especially P) with DOC</td>
<td>Fe concentration</td>
<td>Lab analysis of water samples</td>
<td>Dillon &amp; Molot 2003</td>
</tr>
<tr>
<td></td>
<td>Contaminants of concern: nutrients, solutes, phosphorus, inorganic and organic nitrogen, and sediments</td>
<td>Helps to establish pre-existing contaminant load in wetland</td>
<td>pH, conductivity, Total Dissolved Solids (TDS), nitrate, total nitrogen, ammonium, total phosphorus, and other contaminants relevant to watershed</td>
<td>Lab analysis of water samples</td>
<td>Knox et al. 2008; Cohen et al. 2015</td>
</tr>
<tr>
<td>Vegetation</td>
<td>Plant species and density</td>
<td>Presence of plants anchors substrate, particular species (such as water hyacinth \textit{[Eichhornia crassipes]} and duckweed \textit{[Lemna minor]}) can uptake heavy metals and other contaminants from water</td>
<td>% plant cover, plant species inventory</td>
<td>Field assessment</td>
<td>Salt et al. 1995; McIntyre 2003</td>
</tr>
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</tr>
<tr>
<td>Soil Characteristics</td>
<td>Soil properties</td>
<td>Affects vegetation growth and water treatment capabilities.</td>
<td>Bulk density, moisture content, texture, organic matter and organic carbon content</td>
<td>Lab analysis of soil samples</td>
<td>Peralta et al. 2013</td>
</tr>
<tr>
<td>Soil composition (peatland vs. mineral)</td>
<td>Soil composition (peatland vs. mineral)</td>
<td>Affects the DOC in wetland (high DOC in peatlands, low in mineral soils)</td>
<td>% composition peatland and mineral soil</td>
<td>Field assessment, lab analysis</td>
<td>Stewardson et al. 2003</td>
</tr>
<tr>
<td>Landscape-level</td>
<td>Catchment land use and landscape position</td>
<td>Determines sediment inputs, organic matter, upstream contaminant retention, hydrologic regime</td>
<td>% proportion of catchment land cover, terrain, distance to water bodies and other wetlands</td>
<td>GIS terrain data and land cover classification</td>
<td>Allan et al. 1997; Dillon &amp; Molot 2003</td>
</tr>
<tr>
<td>Wetland characteristics</td>
<td>Spatial factors: size, shape and position Temporal factors: changes in environmental conditions (i.e. climate) and land use</td>
<td>Size, shape, terrain of wetland Long-term climate and weather data, historical air photos</td>
<td>GIS, field based measurements, long-term regional datasets, air photo archives</td>
<td>Cohen et al. 2015</td>
<td></td>
</tr>
</tbody>
</table>
CONCLUSIONS

Empirical data and new methods are required for robust EIA development and decision-making. The rapid pace of natural resource development globally, matched with the progress of peer-reviewed science has created a gap in knowledge transfer. Lessons from landscape ecology and ecosystem services provide better spatio-temporal context and multidisciplinary approaches for assessing ecological stressors and processes with an array of tools that are easily implementable and have a defensible foundation in scientific literature. Lessons, tools and data from LE and ES can provide a way to simplify information for better use by regulators, decision-makers and resource managers, and more accurately and objectively account for the impacts of development decisions that benefit and impact its citizenry.

Here I examined three feasible opportunities to improve EIA and CEA practice by better integrating LE and ES concepts into the process. First, I established the fundamental importance of the spatial extent and its far-reaching effects on the entire EIA process and subsequent outcomes. Secondly, I showed how aerial images and historical datasets help to discern a more valid baseline of ecosystem components. Finally, I demonstrated how a regulating ecosystem service framework can be reasonably incorporated to capture processes previously considered too complicated for consideration in individual project reviews and thus ensure accurate scoping of project impacts.

The role of EIA, and increasingly CEA, will become increasingly important and criticized as social and political tensions continue to increase between environmental protection and the desire to promote economic growth through natural resource development,. In an era where environmental regulation and management can be uncertain and shifting, EIA remains a constant feature, and one of the few avenues for broad stakeholder input into development
planning and decision-making. More robust and transparent integration of biophysical science into assessment will help to promote the validity of EIA work and support public and proponent confidence in the assessment, review and approval process. The conceptual approaches of LE and ES provide useful insights into improvements in EIA scoping and help to build better connections between scientific research, EIA practice and decision-making.
III: Monitoring long-term responses of boreal wetlands to road disturbances using the Landsat archive

INTRODUCTION

Wetlands provide many different ecosystem services (ES) and play a key role in regulating water quality (i.e. nutrient uptake, biogeochemical processing, and sediment retention) and water quantity (i.e. groundwater recharge, maintenance of hydrological connectivity, and runoff and floodwater retention) across a diversity of landscapes (Zedler 2003; Vanderhoof et al. 2016). Locally, wetlands are thought of as the “kidneys” of the landscape (Mitsch & Gosselink 2007). Globally, the provision and delivery of wetland ecosystem services from northern boreal areas (i.e. north of 45° N) are particularly critical as they are estimated to store approximately 50% of the world’s belowground organic carbon (Tarnocai et al. 2009) and provide large shallow aquifers for surface water storage (Morris et al. 2011; Waddington et al. 2015). Approximately 20% of the world’s wetlands are located in the boreal forests of Canada (Webster et al. 2013). In Alberta’s boreal, wetlands (primarily peatlands) occupy approximately 50% of the land surface and are integral to the regional hydrology (Pelster et al. 2008).

Hydrologic processes are fundamental to wetland functioning (Mitsch and Gosselink 2007) and influences biogeochemical cycling, soil formation, and vegetation composition (Haapalehto et al. 2014). As such, disruption of hydrological connectivity due to anthropogenic disturbance alters the delivery/regulation of key ES (McLaughlin and Cohen 2013). In peatlands, several important hydrological connections are susceptible to disruption from anthropogenic disturbance. Internal lateral and vertical flows as well as lateral flows connecting peatlands to the surrounding landscape (Eppinga et al. 2009; Whitfield et al. 2009) can potentially alter wetland
function. While the impact of disrupting these hydrological connections are complex and not well understood, a wide range of important wetland ES might be impacted (Chapter 2).

**Potential Impacts of Roads on Wetlands**

Road networks are continuously expanding throughout boreal areas potentially impacting the hydrologic dynamics of wetlands (Williams et al. 2013). Roads and other linear features (i.e. seismic lines, pipeline right-of-ways) are estimated to account for 80% of boreal anthropogenic disturbance (Pasher et al. 2013). Despite this, few studies have investigated the impacts of roads on wetland hydrological connectivity (Webster et al. 2013). Historically, roads were built through peatlands atop raised, compacted mineral beds to help withstand flooding and prevent road subsidence (Graf 2009). Building such raised linear infrastructure through peatland complexes can alter drainage patterns, disrupt hydrological connectivity, and cause peat compaction (Young et al. 2017). Contemporary road construction (in terms of road placement and associated practices) has improved in boreal areas and now involves culverts to maintain lateral hydrologic flow. However, even culverts can still impact natural drainage as they may become blocked by sediment or beaver dams or installed in inappropriate locations (Mader 2014; Bocking et al. 2017). Furthermore, the effects of roads can be very persistent on the landscape (Chapter 2), lasting long after their construction (Webster et al. 2013). This legacy may be particularly so for older roads originally built with improper placement, insufficient mitigation, and overall more intrusive construction practices. Therefore, understanding the long-term hydrologic impacts of roads, and potential patterns of wetland recovery, is critical.

When roads alter natural drainage patterns and reduce hydrological connectivity, both desiccation and flooding can result (Jeglum 1975; Kahklen & Moll 1999; Patterson & Cooper 2007; Bocking et al. 2017). Desiccation (typically in localized areas down-gradient from roads)
increases the depth to water table, alters plant growth and species composition, and reduces future peat accumulation (Young et al. 2017). Enhanced oxygen availability causes rapid aerobic decomposition of previously saturated peat, releasing CO₂ into the atmosphere (Waddington et al. 2015, Young et al. 2017). Flooding (typically in areas just up-gradient from roads) lowers depth to water table as well as oxygen and soil temperatures creating difficult conditions for plant growth (Bocking 2017).

Developing a wetland monitoring solution

The hydrologic variability of wetlands makes effective monitoring programs difficult to plan and execute (Toyra & Pietroniro 2005). The results of field visits can vary with season, recent precipitation, snow pack, and timing of melt, as well as the expertise of the assessor. To further complicate monitoring, wetland complexes situated in remote and/or expansive areas are difficult and expensive to access for systematic field assessments (Toyra & Pietroniro 2005). For large and remote areas such as northern peatland complexes in the boreal forest of Canada, the utilization of remote sensing (RS) methods present a great opportunity to employ a standardized, informative, low-cost approach to monitoring. Remote areas, where competing pressures for natural resources and wetland conservation likely co-exist, are optimal locations to employ remote sensing wetland monitoring and support multiple objectives.

To fulfill this need for Canada, as well as improve upon wetland monitoring using remote sensing (RS) more generally, I examine the effects of roads on wetland areas using a 27-year RS time series. First, I evaluate long-term responses to roads placed adjacent to several different types of wetlands with a replicated, multi-wetland analysis. As part of this, I aim to evaluate a suite of remote sensing indices along with a series of hypotheses regarding potential wetland recovery trends. To rigorously control for variations in temperature and precipitation, I used a
paired design to create controls (pairing wetlands close and far from roads). Second, I also investigate finer spatiotemporal trends within in a single wetland to examine localized hydrologic changes.

Previous literature on hydrological connectivity of wetland/peatland complexes provides several expectations regarding the effects of roads (Belyea and Baird 2006; Waddington et al. 2015; Bocking et al. 2017) synthesized in Figure 3.1. If the effects of road building are detectable using remote sensing, I hypothesize that wetlands may eventually recover to pre-disturbance conditions, but with delay (Hypothesis 1 - time-lag recovery). Such a delay could either be caused by strong memory effects and/or recovery of hydrological connectivity. Some wetlands may not visibly recover, however, indicating a potential regime shift to wetter OR drier conditions than before road building (Hypothesis 2) potentially as a result of disruptions to hydrological connectivity. My null hypothesis is that the effects of road building are either not detectable with RS analysis or, alternatively, that peatlands are complex and highly adaptive (Belyea & Baird 2006) and thus able to compensate for disruptions in hydrological connectivity (Hypothesis 3).

Finally, as part of my examination of a single wetland, I expect to be able to distinguish RS trends in wetland areas immediately adjacent to the road based upon localized hydrologic gradients. Similar to Kahklen & Moll (1999), Wang et al. (2010), and Bocking et al. (2017), I expect the up-gradient side will be subject to flooding (Figure 3.1 in blue) whereas the down-gradient area will experience desiccation (Figure 3.1 in brown).
Figure 3.1. Depiction of the hypothesized outcomes of wetland hydrological response to road building disturbance. Outcomes with increasing wetness (flooding) are positive on the y-axis and shown as shades of blue/green. Conversely, decreasing wetness (desiccation) is shown in shades of brown (negative y-axis values). The red dashed vertical line refers to the road building event and the x-axis represents the time before/after the disturbance. Hypothesis 1 (dotted black line) refers to a null change outcome where either roads have limited effect on wetlands or changes are not detectable using RS. Hypothesis 2 refers to an initial change in wetland RS indices, followed by a return to pre-disturbance levels (time-lag with recovery). Hypothesis 3 is a regime shift or permanent change to wetter or drier conditions over a longer time-period.
Evaluation of wetland trends, as well as the rigour of the tools used in their assessment, becomes increasingly important as access and exploration of the boreal forests of Canada increases. As development continues in the boreal forest in northern Alberta, understanding the impact of disturbances such as roads on wetlands, as well as the time frame for recovery, is critical. Based upon the approved oil and gas development in Alberta by 2005, an estimated 30,000 km of access roads is needed to extract these resources (Schneider & Dyer 2006). Despite the importance of boreal wetlands regionally and globally, limited research has been conducted on the magnitude and duration of anthropogenic impacts such as roads on hydrological connectivity (Ferone & Devito 2004, Whitfield et al. 2009; Wang et al. 2010). This work provides a way forward and will achieve three main objectives:

1. Determine the time lags over which changes in wetland RS indices are detectable following road building.
2. Determine if change detection is impacted by initial conditions (i.e., RS indices for different wetland types).
3. Characterize localized spatial patterns of change following road construction in upslope and downslope gradients using wetland RS indices.

METHODS

Study Area

The Central Mixedwood Natural Subregion (CMNS) in the boreal forest region of Northern Alberta contains heterogeneous glacial deposits (ranging from 20-240 m in depth) which have created complex surface hydrology resulting in a combination of mixedwood forest, mineral wetlands, and peatlands (Smerdon et al. 2008; Ketcheson et al. 2016). Peatlands (including bogs and fens) are estimated to occupy nearly 50% of the surface area of the CMNS.
The sub-humid climate in this area ranges from cold winters (-16.7°C monthly average in January) to warm summers (16.3°C in July) from 1970 to 2000 (Devito et al. 2005). The majority of precipitation (60%) occurs as short, intense rain events during June, July, and August and snowfall (25% of precipitation) usually covers the ground from early November until late April (Devito et al. 2005). The potential evapotranspiration slightly exceeds precipitation based upon yearly totals (Ketcheson et al. 2016).

The geology, climate, and history of natural resource development in the CMNS make it an ideal location to study the impact of road building on wetlands. The deep glacial sediment deposits, flat topography, and cooler temperatures result in a large amount of water stored and distributed in wetlands (Devito et al. 2005; Devito et al. 2012). Prior to the 1960s, limited anthropogenic disturbance occurred in the CMNS because of its remote location and inaccessibility. However, since the inception of natural resource extraction (particularly petroleum) in the 1960s, the scale of development (both in terms of spatial extent and total production) has expanded considerably over 60 years (Boutin & Carpenter 2017). To facilitate this increase in development, an extensive network of roads has been built to explore and extract resources in the CMNS.

**Overview of landscape-level monitoring design**

To meet the objectives of this study, I used a 27-year time series of Landsat satellite imagery to evaluate biophysical RS indices proven useful in long-term wetland monitoring. I compared these indices among different types of wetlands in areas impacted by road building. To identify wetlands for analysis, I performed a baseline wetland classification in 1985 and intersected subsequent road-building as of 2012. Then, I analyzed a suite of wetland RS indices for each wetland (annually from 1985-2012). To compare multiple wetlands (Objectives 1 and 2)
before and after road construction, I used a before/after control/impact (BACI) statistical design to ascertain multi-wetland trends across the region. To rigorously control for variations in temperature and precipitation, I used a paired design to create controls (pairing wetlands close and far from roads). To verify BACI trends and further examine spatial and temporal pixel-based change at a single wetland (Objective 3), I used a break for additive and seasonal trends (BFAST) monitoring approach. The results of multiple and single wetland analysis were combined to gain insight into my regime shift/recovery hypotheses (Fig. 3.1) and to determine if a spatial pattern of trends was evident within a wetland in areas upstream (expected to become wetter) vs. downstream (expected to become drier) of road construction. Next, I explain each of these steps in more technical detail.

**Wetland Identification, Selection, and Comparison**

*Baseline Wetland Identification*

To locate a baseline of different wetland types prior to road construction, I used Landsat imagery from 1985 to represent initial ‘baseline’ conditions, the earliest year for which imagery was available. Based on historical monthly precipitation from nearby Fort McMurray, conditions around the date of image acquisition were within a typical range and did not represent anomaly conditions (Government of Canada 2018). Building on the strengths of prior studies evaluating different remote sensing indices for wetland identification and monitoring (Crist & Ciccone 1984; Baker et al. 2006; Wright & Gallant 2007; Xie et al. 2008; Kayastha et al. 2012; Fickas et al. 2016), an orthogonal tasseled cap transformation (TC) was performed on the 1985 image. TC uses six Landsat bands (1-5 and 7) to calculate three indices – brightness, greenness and wetness using methods first proposed by Crist and Ciccone (1984). The three indices of the TC are associated with changes in modified/anthropogenic surfaces (brightness), vegetation (greenness),
and soil moisture (wetness) (Xie et al. 2008, Baker et al. 2007). The TC was performed using the ‘RStoolbox’ package in R (version 3.4.2) (Leutner et al. 2018).

To create a consistent method for wetland identification across the region, I conducted an unsupervised classification with the ‘RStoolbox’ package in R (Leutner et al. 2018) based upon the three TC output bands. The classification was conducted on the entire CMNS region and utilized k-means clustering with the Hartigon-Wong algorithm. The output contained 10 classes based upon a sample of 10,000 random pixels and 100 full cluster analysis iterations. I chose 10 classes for the unsupervised classification based on contemporary land-use classifications in the same area (Ducks Unlimited 2018). Of the 10 total classes, 3 consistent wetland classes were identified to represent “baseline wetland types” for the purposes of this research. To test the accuracy of the classification, an exploratory map comparison was conducted (see Appendix A). Yet for the baseline imagery, a severe lack of supporting data (e.g., either a historical wetland classification or high spatial resolution imagery from 1985) for this region prevented a typical ground verification. Thus, the approach taken for the 1985 classification was applied to a 2012 image and evaluated using the Alberta Environment and Parks (AEP) Merged Wetland Inventory (AEP 2018) dataset. Further details on the map comparison are provided in Appendix A.

Identification of Wetlands Subject to Road Development

To identify baseline wetlands subsequently impacted by road development, I intersected baseline wetlands with roads data from the 2012 Alberta Biodiversity Monitoring Institute (ABMI) Human Footprint Inventory (HFI) (v.3, 2012) in ArcGIS (version 10.5). The HFI combines several different geospatial datasets from the Alberta government and industry to map temporary or permanent residential, recreational and industrial land uses and infrastructure. Additional details on the ABMI HFI dataset is provided in Appendix B. I eliminated wetlands
within 1000 m of any other anthropogenic disturbance using additional layers of the HFI (ABMI 2012) to avoid confounding road impacts with other forms of development. Each selected wetland was also visually assessed in 1985, 1993, 2000, 2006, and 2012 using a combination of Landsat and Google Earth imagery. This additional measure was taken to verify the location of roads (and if necessary, make manual adjustments), as well as verify lack of nearby disturbances (within 1000 m) such as forest harvest or fire. Wetlands not meeting these criteria were eliminated from analysis.

In the multi-wetland analysis, it was necessary to control for potential opposing gradient effects (flooding in up-gradient areas, desiccation in down-gradient areas). Thus, I only selected wetlands with roads intersecting the wetland edge (and not the centre) on the down-gradient side. This positioning helped ensured greater consistency in the direction of change trajectories following road disturbance. Elevation was assessed for each wetland site using a 25-m Digital Elevation Model (DEM) raster provided by AltaLis LTD compiled from 1:60,000 aerial photography from 1955 to 1996 (AltaLis, 2018). The 25-m DEM was further used in the single wetland analysis to compare up- and down-gradient impacts of roads within in a wetland complex (additional details provided in methods for Objective 3).

Identification of Control Wetlands

Control wetland sites were identified (for use in multi-wetland level analysis) to account for yearly fluctuations in temperature, snowpack, and precipitation and long-term shifts in climate within the region. Control wetlands representing wetland types found in the baseline 1985 classification were selected to ensure comparability of controls with wetland characteristics at impact sites. Then, each wetland impacted by road development was paired with a control, subject to the following requirements: 1) both control and impact were within 5 km of each
other, and 2) controls were >1000 m from any HF disturbances or roads. Potential control sites were visually assessed using 1985, 1993, 2000, 2006, and 2012 imagery to eliminate those with anthropogenic disturbances (logging or signs of forest fire). Any control sites not meeting these requirements were eliminated and when multiple controls met these requirements, the wetland closest in size was selected.

**Long-term Wetland Remote Sensing**

*Landsat Best Available Pixel (BAP) database*

Once wetland selection was complete, I measured change in RS indices over a 27-year time series using the Best Available Pixel (BAP) Landsat imagery database compiled by Hermosilla et al. (2015a, 2015b, 2016). The BAP was compiled using a pixel-based image compositing approach from available Landsat Thematic Mapper (TM) and Enhanced Thematic Mapper Plus (ETM+) images spanning 1 August ± 30 days from 1985 to 2012 to correspond to the growing season and reduced cloud cover (Hermosilla et al. 2016). All imagery was Level-1 Terrain-Corrected (L1T) products and atmospheric correction surface reflectance values were derived using the Landsat Ecosystem Adaptive Processing System (LEDAPS) algorithm (Hermosilla et al. 2016). Clouds and shadows were detected and masked using the Fmask algorithm (Zhu and Woodcock 2012; Hermosilla et al. 2016). To create the BAP, individual pixels were selected for composite images using pixel-scoring functions (described in White et al. 2014) to create a seamless longterm high quality image. Additional details of this procedure can be found in Hermosilla et al. (2015a, 2015b, 2016) and some minor constraints of the database of relevance to this research are discussed further in *Limitations.*

*Tasseled cap transformations*
The use of a TC on a dense time-series (such as annual Landsat imagery) is an effective way to detect wetland (particularly forested wetland) change (Baker et al. 2006; Ordoyne & Friedl 2008; Kayastha et al. 2012; Li et al. 2015). Although different TC adaptations have been developed for each Landsat sensor (Crist and Cicone 1984), in the BAP dataset, the different sensors were standardized for comparison across the temporal range. As such, I used a single-set of TC coefficients optimized for use with the BAP archive to ensure consistency (Table 3.1; Hermosilla pers. Comm.).

Table 3.1. Tasseled cap transformation coefficients used for cross TM and ETM/ETM+ sensor consistency.

<table>
<thead>
<tr>
<th>Index</th>
<th>Band 1</th>
<th>Band 2</th>
<th>Band 3</th>
<th>Band 4</th>
<th>Band 5</th>
<th>Band 7</th>
</tr>
</thead>
<tbody>
<tr>
<td>Brightness</td>
<td>0.3561</td>
<td>0.3972</td>
<td>0.3904</td>
<td>0.6966</td>
<td>0.2286</td>
<td>0.1596</td>
</tr>
<tr>
<td>Greenness</td>
<td>-0.3344</td>
<td>-0.3544</td>
<td>-0.4556</td>
<td>0.6966</td>
<td>-0.0242</td>
<td>-0.2630</td>
</tr>
<tr>
<td>Wetness</td>
<td>0.2626</td>
<td>0.2141</td>
<td>0.0926</td>
<td>0.0656</td>
<td>-0.7629</td>
<td>-0.5388</td>
</tr>
<tr>
<td>Fourth</td>
<td>0.0805</td>
<td>-0.0498</td>
<td>0.1950</td>
<td>-0.1327</td>
<td>0.5750</td>
<td>-0.7775</td>
</tr>
<tr>
<td>Fifth</td>
<td>-0.7252</td>
<td>-0.0202</td>
<td>0.6683</td>
<td>0.0631</td>
<td>-0.1494</td>
<td>-0.0274</td>
</tr>
<tr>
<td>Sixth</td>
<td>0.4000</td>
<td>-0.8172</td>
<td>0.3832</td>
<td>0.0602</td>
<td>-0.1095</td>
<td>0.0985</td>
</tr>
</tbody>
</table>

**Change detection and time series analysis**

To select the appropriate method to detect and monitor changes in wetland areas using remote sensing, it is imperative to consider the type, duration, and spatial extent of the disturbance/change (Coppin et al. 2004). As such, I utilized different methods of digital change detection and time series analysis to complete each objective. Prior to analysis of multiple (Objective 1 and 2) or single wetlands (Objective 3), it was critical to establish the correct before/after timeline of road building in each site. To identify the correct year of disturbance, I used a change detection method, Change Vector Analysis (CVA). For multiple wetland analysis (Objective 1 and 2), a Before/After Control/Impact (BACI) design was implemented using TC
indices to determine the type and temporal patterns of wetland change. For single wetland analysis (Objective 3), pixel time-series surrounding a road disturbance were analyzed using the Breaks For Additive and Seasonal Trend (BFAST) monitoring method to confirm results of the multiple wetland analysis and provide insight into the spatial dynamics of change. Technical details of the analysis are provided in the relevant sections below.

**Developing the timeline: Change vector analysis to find the year of road disturbance**

Due to the large number of wetlands, I employed a change detection method to identify year of road building using the “RStoolbox” package in R (Leutner et al. 2018). CVA determines the direction and magnitude of pixel change in multispectral space (i.e. utilizes change in all image data layers) (Lambin & Strahlers 1994; Lambin & Elhrich 1997). CVA sets a change threshold to differentiate between true landscape changes (such as changes in land cover), from variability due to radiometric, atmospheric and other subtle changes (Hame et al. 1998, Johnson & Kasischke 1998). Setting a threshold is particularly important in dynamic ecosystems such as wetlands where variation occurs seasonally and among years.

I used CVA to compare brightness and greenness TC spectral bands between 1985 and each subsequent annual image. Outputs included two bands, both a magnitude of change and a direction of change band. The magnitude of change is expressed as the value in exceedance of the threshold. Here, the change threshold was set as 2x the median (non-zero) value. The direction of change is calculated as an angle from 0 to 360°. The combination of these two outputs indicates substantial pixel change relative to elsewhere in the image. Taken together, this output provides information on how pixel values changed (i.e. an increase in brightness and a decrease in greenness) associated with road building.
The specific angle of change associated with road building was determined through visual assessment of CVA results at known road features. CVA values within the range associated with a road-building disturbance were then extracted, using the “RStoolbox” package in R (Leutner et al. 2018), for road shapefiles in each year. Year of road building was identified as the first exceedance of the CVA threshold within the angle of change associated with road building. An example of this process for one wetland is shown below in Figure 3.2. Year of road construction was also manually verified using Landsat.
Figure 3.2. An example using CVA to identify road disturbances. The purple polygon represents an example wetland area. Panel 1 (left) shows a true-color Landsat image from 1993 and an intact wetland polygon prior to road construction. Panel 2 (middle) provides the results of CVA between 1985 and 1994 (the first incidence of CVA results within an annual chronosequence beginning in 1986). The yellow (low) to red (high) spectrum in Panel 2 represents the magnitude of CVA results associated with road building (increases in TC brightness, decrease in TC greenness). Panel 3 (right) demonstrates the 1994 true-color Landsat image where the road is visible thus confirming the results of the CVA analysis.
**Objective 1 & 2: Statistical Analysis of Multiple Wetland Trends Using BACI**

Following the novel use of Before/After, Control/Impact (BACI) for RS analysis by Meroni et al. (2017), I applied the approach to compare TC brightness, greenness, and wetness indices following road building across multiple wetlands over the long-term. The BACI design is commonly used in ecology to assess the impact of stressors but has been seldom applied to remote sensing. I performed multiple BACI iterations using smaller temporal windows within the 27-year TC dataset to identify temporal trends (Objective 1) and to determine if trends differed among wetland types (Objective 2). To do so, BACI models evaluated TC indices in windows flanking the years before and after construction, ranging from 1, 3, and 5 years pre- and post-disturbance. Longer duration models (e.g., 5-year) included data from shorter duration (e.g., 1 and 3 year) model iterations.

I used a linear mixed effects (LME) model from the “lme4” (Bates et al. 2015) and “lsmeans” (Lenth 2016) packages in R studio to complete BACI analysis as outlined in Schwarz (2015). The LME model treats the period (Before vs. After road disturbance), the class of each wetland (Control vs. Impact), and the interaction between period and class (the BACI effect) as fixed effects. Variation among wetlands, annual variation, and interaction between class and year were random effects. The least-squares means of period * class interaction was used to calculate the BACI contrast using p = 0.05. The BACI contrast is calculated as:

\[
\text{BACI contrast} = \mu_{CA} - \mu_{CB} - \mu_{IA} + \mu_{IB} \quad (\text{Eq. 1})
\]

Where \( \mu \) is the mean TC values within each period and class treatment (ex. control/after), C is for control sites, I is for impact sites, A is for after impact, B is for before impact. A negative contrast indicates the variable increased more at impact sites than in the controls. A positive contrast indicates the variable is greater in the control sites and has decreased in the impact sites.
While the units of the BACI contrast match the units of the input variable, in order to better contextualize TC changes, I normalized BACI contrasts as percent change (%) as in Meroni et al. (2017).

Objective 3: Spatial pattern of flooding/drying within a single wetland

To further investigate road impacts at a finer-scale, one impacted wetland was assessed in greater spatiotemporal detail following road construction in 1993 until 2002 (when nearby forest harvest became evident). Within a 2 km x 2 km wetland area, annual TC wetness values at individual pixels were assessed as distinct time series using the Breaks for Additive and Seasonal Trend (BFAST) monitoring method with the “bfast” package in R Studio (Verbesselt et al. 2011). TC wetness was selected for further analysis for two reasons. First, wetness corresponds with soil moisture and water table depth (Ordoyne & Friedl 2008) and may provide a useful proxy for measuring hydrological disruption. Second, wetness breakpoints have been widely observed in disturbed forested ecosystems (Oeser et al. 2017). BFAST monitoring was developed for real-time disturbance detection in remote sensing time series. For my analysis, the time series was split into a stable period prior to road construction (1985 to 1992) as well as a monitoring period following road construction (1993 to 2002). The time series at each pixel was tested for structural breaks whereby an OLS regression model (Eq. 2) was first fit to the stable period using a standard linear regression model:

\[ Y_t = x_t \beta + \epsilon_t \]  

(Eq. 2)

and then post-disturbance values were assessed for breakpoints deviating from the historical period using a moving sums of the residuals (MOSUM) approach. If TC wetness values post-disturbance \((t = n+1, \ldots, N)\) do not deviate from the stable period, the MOSUM will be close to zero. If the MOSUM deviates systematically from zero and surpasses a boundary \((\leq 5\%)\)...
probability of occurring in the stable period), a structural break is identified (see Verbesselt et al. 2012 for additional details). The magnitude and direction (+/- change in indices) is determined from the difference between the medians of the stable and monitoring periods, along with the year of break detection. To address my last objective and determine if there were contrasting effects of road building based on flow gradients, BFAST breaks were ascertained separately within up- and down-gradient areas of the wetland.

RESULTS

Tasseled-cap is useful for identifying historical wetland areas

The unsupervised classification of the 1985 baseline Landsat imagery produced three classes which captured transitional wetland ecosystems situated between open water and upland forest classes. These three wetland classes ranged in TC brightness, greenness and wetness yet showed consistent spectral signatures within each class: high wetness low greenness (HWLG), medium wetness low greenness (MWLG), and low wetness high greenness (LWHG). Based upon the original classification, the HWLG sites are most similar in TC values to open water, whereas the LWHG sites are the most similar in TC values to upland forest of the three wetland types, with the MWLG sites representing an intermediate between the two classes. After using a combination of GIS and visual assessment methods, 18 HWLG, 23 MWLG, and 45 LWHG wetlands were used for further analysis. A lack of high spatial resolution imagery or land-cover map for the region in mid-1980’s prevented a formal, robust accuracy assessment yet an exploratory map comparison is provided in Appendix A.

Change vector analysis identified the correct year of road building in 77% of cases

For 86 sites (HWLG, MWLG, and LWHG combined), the accuracy of using CVA to identify the year of road building in wetland areas was 77%. The accuracy within ± 1 year of
road building increased to 93% (Figure 3.3). The largest difference between the year determined by CVA compared to visual interpretation was 3 years (3 sites). Of the sites incorrectly identified, 8% were too early whereas 15% were detected late, after the year of road building.

![Graph showing the comparison between the estimated year of disturbance based upon CVA and by visual interpretation. The accuracy of this technique for determining the year of disturbance was 77% (66 sites). The accuracy within ±1 year was 93% (83 sites). Black points located on the 1:1 horizontal line represent 100% accuracy, dark grey points represent ±1 year, and light grey points represent ±2 years or more.]

**Figure 3.3: Using CVA to determine the year of road construction for 86 sites.** Shown is a comparison between the estimated year of disturbance based upon CVA and by visual interpretation. The accuracy of this technique for determining the year of disturbance was 77% (66 sites). The accuracy within ±1 year was 93% (83 sites). Black points located on the 1:1 horizontal line represent 100% accuracy, dark grey points represent ±1 year, and light grey points represent ±2 years or more.

**Objective 1: Trends in RS indices were detected three and five years after road construction**

Several significant BACI Contrasts (BC) were detected three and five years after road construction (Table 3.2). Percent change is also shown in Figure 3.4. For all wetland types and all indices, there were no significant BC results detected within one year of road building.
However, it was not possible to evaluate trends 10-years before and after construction due to sample size limitations.

Trends were detected for TC mean wetness (4 models) and mean brightness (3 models) (Table 3.2); however there were no significant BC results for TC mean greenness in any BACI iteration. Recall an increase in indices results in a negative BC. Thus, to clarify the trajectory of BC changes, I have shown percent change (Fig. 3.4) as the actual change in the TC indices (i.e. increasing brightness associated with a negative BC).

Focusing on the significant BC results, increases were detected for TC mean brightness in the 3-year (MWLG) and 5-year (HWLG and MWLG) models. Percent change in mean brightness ranged from an increase in just over 6% (±3 SE) five years after road construction in HWLG and an initial increase of 9% (±4 SE) at three years to 8% (±3 SE) in 5-year models for MWLG (Figure 3.4). Decreasing TC wetness was measured in the 3- and 5- year models in both the HWLG and MWLG wetlands. In HWLG wetlands, an initial decrease of 11% (±4 SE) was detected after three years and a decrease of 9% (±2 SE) was detected after five years. Similar trends were detected in MWLG wetlands, similar trends were detected, a decrease of 12% (±3 SE) in 3-year and 10% (±3 SE) after 5-year models (Figure 3.4).

**Objective 2: Trends detected in “wetter” wetlands**

Within wetland types (see Table 3.2), significant BC results for TC indices were detected in HWLG (3 models; column 1) and MWLG (4 models; column 2). There were no significant results for LWHG (column 3). Further evidenced by Figure 3.4, percent change in mean brightness, greenness, and wetness after road construction was close to 0% for LWHG, indicating very little variation in TC values after road building. LWHG sites had lower initial TC wetness and higher greenness values compared to MWLG and HWLG sites. Based upon the
original classification, of the three wetland types, LWHG’s TC values are the most similar to upland forest.
Figure 3.4: Percent change in mean tasseled cap (TC) indices at one, three and five years following road construction relative to control wetlands. The relative standard errors (SE) are shown for each result (see Table 2). Decreases in wetness and increases in brightness were significant at 3 and 5 years for HWLG and MWLG. Changes in greenness values were insignificant for all wetland types. No TC indices were significant for LWHG. Percent change was measured as % change as compared to the range of values of TC indices for adjacent control wetlands at the same year following road building. Significant differences indicated by * (for p values <0.05).

Table 3.2. Impact of roads on TC indices measured in adjacent wetlands at 1-, 3-, and 5-years following construction as determined using BACI linear mixed effects models for three wetland types (HWLG, MWLG, and LWHG). BACI Contrast (BC) is the difference (control vs. impact) between the mean differences (after vs. before) (see full text for formula). For brightness values, a negative BC indicates an increase in brightness in the road-impacted sites whereas a positive BC for wetness values represents a decrease in wetness in road-impacted sites. Thus, to help interpretation, percent change of the BC and the associated standard error are also reported in bold and shown visually in Fig 4. BC results with p values <0.05 (where the null hypothesis of no change was rejected) are in green. The significance level is indicated by * for p-values <0.05 and ** for p-values of <0.01.

<table>
<thead>
<tr>
<th>Years Post-Road</th>
<th>BACI model parameters</th>
<th>HWLG</th>
<th>MWLG</th>
<th>LWHG</th>
</tr>
</thead>
<tbody>
<tr>
<td>RS Indices 1: Mean TC brightness</td>
<td>BACI Contrast (BC)</td>
<td>-43.94</td>
<td>-92.98</td>
<td>-12.41</td>
</tr>
<tr>
<td>1 Year</td>
<td>BC - Standard Error</td>
<td>70.24</td>
<td>65.64</td>
<td>53.17</td>
</tr>
<tr>
<td></td>
<td>p-value</td>
<td>0.54</td>
<td>0.16</td>
<td>0.82</td>
</tr>
<tr>
<td></td>
<td>Percent Change (%)</td>
<td>-6.13 (±9.8)*</td>
<td>-8.27 (±5.84)</td>
<td>-0.93 (±3.99)</td>
</tr>
<tr>
<td>3 Year</td>
<td>BACI Contrast (BC)</td>
<td>-73.8</td>
<td>-104.87</td>
<td>-10.53</td>
</tr>
<tr>
<td></td>
<td>BC - Standard Error</td>
<td>50.5</td>
<td>39.48</td>
<td>32.30</td>
</tr>
<tr>
<td></td>
<td>p-value</td>
<td>0.15</td>
<td><strong>0.0085</strong></td>
<td>0.74</td>
</tr>
<tr>
<td></td>
<td>Percent Change (%)</td>
<td>-6.64 (±4.54)</td>
<td>-9.34 (±3.52)**</td>
<td>-0.76 (±2.32)</td>
</tr>
<tr>
<td>5 Year</td>
<td>BACI Contrast (BC)</td>
<td>-69.0</td>
<td>-86.35</td>
<td>-14.85</td>
</tr>
<tr>
<td></td>
<td>BC - Standard Error</td>
<td>34.65</td>
<td>31.41</td>
<td>25.74</td>
</tr>
<tr>
<td></td>
<td>p-value</td>
<td><strong>0.048</strong></td>
<td><strong>0.0063</strong></td>
<td>0.56</td>
</tr>
<tr>
<td></td>
<td>Percent Change (%)</td>
<td>-6.21 (±3.12)*</td>
<td>-7.69 (±2.80)**</td>
<td>-0.99 (±1.71)</td>
</tr>
<tr>
<td>RS Indices 2: Mean TC wetness</td>
<td>BACI Contrast (BC)</td>
<td>69.45</td>
<td>80.41</td>
<td>15.57</td>
</tr>
<tr>
<td>1 Year</td>
<td>BC - Standard Error</td>
<td>45.86</td>
<td>42.69</td>
<td>23.75</td>
</tr>
<tr>
<td></td>
<td>p-value</td>
<td>0.14</td>
<td>0.064</td>
<td>0.51</td>
</tr>
<tr>
<td></td>
<td>Percent Change (%)</td>
<td><strong>11.42 (±7.54)</strong></td>
<td><strong>11.80 (±6.27)</strong></td>
<td><strong>2.17 (±3.31)</strong></td>
</tr>
<tr>
<td>3 Year</td>
<td>BACI Contrast (BC)</td>
<td>79.44</td>
<td>86.57</td>
<td>12.68</td>
</tr>
<tr>
<td></td>
<td>BC - Standard Error</td>
<td>28.39</td>
<td>25.98</td>
<td>14.28</td>
</tr>
<tr>
<td></td>
<td>p-value</td>
<td><strong>0.006</strong></td>
<td><strong>0.001</strong></td>
<td>0.38</td>
</tr>
<tr>
<td></td>
<td>Percent Change (%)</td>
<td><strong>11.30 (±4.04)</strong>**</td>
<td><strong>11.62 (±3.49)</strong>**</td>
<td><strong>1.58 (±1.78)</strong></td>
</tr>
<tr>
<td>5 Year</td>
<td>BACI Contrast (BC)</td>
<td>78.52</td>
<td>70.94</td>
<td>16.40</td>
</tr>
<tr>
<td></td>
<td>BC - Standard Error</td>
<td>17.38</td>
<td>19.60</td>
<td>12.30</td>
</tr>
<tr>
<td></td>
<td>p-value</td>
<td><strong>0.0001</strong></td>
<td><strong>0.0003</strong></td>
<td>0.18</td>
</tr>
<tr>
<td></td>
<td>Percent Change (%)</td>
<td><strong>9.41 (±2.08)</strong>**</td>
<td><strong>9.53 (±2.63)</strong>**</td>
<td><strong>1.49 (±1.12)</strong></td>
</tr>
<tr>
<td>RS Indices 3: Mean TC greenness</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>
### Objective 3: Single-wetland analysis most clearly confirms the timing of change, not the direction of hydrologic impact

Analysis of one wetland area was conducted to confirm the results of the BACI models and to provide insight into the spatiotemporal dynamics in TC wetness. A chronosequence of changes in TC wetness before and after road disturbance is shown in Figure 3.5. The bisecting road is first visible in 1993 as dark brown (very low wetness). The TC wetness appears to fluctuate in 1994 and 1995 and then transition to a consistently drier state (lower wetness, lighter blue color) from 1996 to 2002. BFAST was used to statistically evaluate the timing, magnitude, spatial distribution, and topographic drivers (i.e. slope, elevation) associated with these dynamics.
Figure 3.5: Change in TC wetness before and after road building. TC wetness is displayed over time from August 1985 to 2002, before and after road construction circa 1993. Dark blue pixels indicate higher TC wetness and dark brown pixels are drier. The darkest linear brown pixels correspond to road disturbance. Little variation in TC wetness was observed prior to road disturbance. Yet in the years after road disturbance, TC wetness appeared to fluctuate in areas adjacent to the road, initially increasing until 1995 but then decreasing through 2002. All images are within 30 days of August 1.
After road construction, structural breaks were detected in the time series for 43% of pixels (Figure 3.6), albeit at different years. Just over half of the structural breaks occurred within the first three years of road building. Approximately 5% of breaks were detected by year one, 17% in year two, and 30% in year three (Table 3.3). Years 4 - 10 each accounted for 2-12% of breaks (totalling ~48% overall). Locations of temporal breaks (Figure 3.6) within the first three years (green) appeared closer to the road whereas subsequent breaks (purple) were more dispersed throughout the area.

Table 3.3. Percentage of pixels with structural breaks after road construction detected using BFAST (at a significance level ≤5%).

<table>
<thead>
<tr>
<th>Years After Road Disturbance</th>
<th>Year</th>
<th>Structural Breaks (%)</th>
<th>Cumulative Structural Breaks (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td>&lt;1</td>
<td>1993</td>
<td>4.98</td>
<td>4.98</td>
</tr>
<tr>
<td>2</td>
<td>1994</td>
<td>17.22</td>
<td>22.20</td>
</tr>
<tr>
<td>3</td>
<td>1995</td>
<td>30.27</td>
<td>52.47</td>
</tr>
<tr>
<td>4</td>
<td>1996</td>
<td>2.89</td>
<td>55.36</td>
</tr>
<tr>
<td>5</td>
<td>1997</td>
<td>12.38</td>
<td>67.74</td>
</tr>
<tr>
<td>6</td>
<td>1998</td>
<td>2.89</td>
<td>70.63</td>
</tr>
<tr>
<td>7</td>
<td>1999</td>
<td>11.01</td>
<td>81.64</td>
</tr>
<tr>
<td>8</td>
<td>2000</td>
<td>9.77</td>
<td>91.41</td>
</tr>
<tr>
<td>9</td>
<td>2001</td>
<td>3.32</td>
<td>94.73</td>
</tr>
<tr>
<td>10</td>
<td>2002</td>
<td>5.27</td>
<td>100.00</td>
</tr>
</tbody>
</table>
Figure 3.6. Temporal distribution of BFAST structural breaks. The year of structural break is shown as green (earlier) to purple (later) in the time-series. Pixel time-series without structural breaks are shown in gray. The road pixels (identified manually in each annual image after disturbance) are shown in white. Early detections occurred directly adjacent to the road as well as in other clustered areas (i.e. left side of the image). Later detections (1997-2002) appear more evenly dispersed.
The year and magnitude of post-disturbance breakpoints varied spatially. Of pixels exhibiting statistically significant trends, negative breaks (decreasing wetness) occurred in 68% of pixels and positive in 32% (Figure 3.7). The highest magnitude negative breaks (shown in red) occurred near and adjacent to the road (Figure 3.7, Panel A). Positive breaks did not appear to be located near the road of interest; however, several other, unrelated linear patterns were observed with positive breaks (Figure 3.7, Panel B).

The greatest range of annual breakpoint magnitude values occurred within the first three years of road construction and then subsequently decreased until 2002 (Figure 3.8). The majority of breakpoints within each year fell below zero (1993, 1996, 1997, 1998, 1999, 2000, 2001 and 2002) with the exception of two years (1994 and 1995, refer back to Figure 3.8). The increasing TC wetness apparent in BFAST magnitude distribution in 1994 and 1995 was also observed in the chronosequence in Figure 3.5.
Figure 3.7. Spatial distribution of the magnitude of BFAST breakpoints. The negative and positive breakpoints were depicted separately (Panel A and B) to identify spatial trends. The left panel (A) shows the distribution of negative breakpoints (decreasing wetness) from light orange (lowest magnitude) to dark red (greatest magnitude). The right panel (B) shows the distributions of positive breakpoints (increasing wetness) from light blue (lowest magnitude) to dark blue (greatest magnitude). The grey represents pixels where no breakpoint was detected (see Figure 3.6 for comparison) or pixels with a structural breakpoint of the opposite magnitude. As in Figure 3.6, the white pixels were identified as roads and are not included in the analysis. In panel A, the greatest magnitude pixels (dark red) were located in the area adjacent to the road. The lower magnitude breakpoints (light orange) were observed more evenly across the area. In panel B, the location of positive breakpoints did not appear to be related to the road but were observed in linear patterns.
Figure 3.8. A violin plot of BFAST breakpoint magnitude in each year following road disturbance. The sample size (number of pixels) is provided in the center of each violin plot. The vertical length of each violin plot represents the range of values within each year. The width of each violin plot represents the density of points that fall at the corresponding magnitude. The greatest range of within year variability was detected in the first three years following road construction (1993, 1994, and 1995). The majority of breakpoints (shown visually as the greatest plot width) of each year fell below zero (indicated by a red dashed line) indicating decreasing wetness, with the exception of 1994 and 1995.
**Trends in water-table depth not affected by up vs down-gradient**

It was more feasible to examine site-specific differences, such as gradient, that might interact with road construction as part of a singular wetland analysis. The road position at the selected wetland site was roughly parallel to the contours (Figure 3.9, Panel A and B) producing an up-gradient area southwest of the road as well as a down-gradient area northeast of the road. Of the significant BFAST trends detected, 43% (910 pixels) were up-gradient and 56% (1198 pixels) were down-gradient. Visually, there was no clear difference between the location of positive and negative breakpoints with gradient position (Figure 3.9, Panel B). The lack of difference over elevation was also confirmed by the similar distribution of positive and negative BFAST monitor results as depicted in Figure 3.10.
Figure 3.9. Comparison of the wetland elevation (left, Panel A) to the distribution of positive and negative BFAST breaks (right, Panel B). The elevation in Panel A ranges from lower elevation (green) to higher elevation (white). A 3-D contour along the north and east side of the wetland is also shown on the border of the map in grey. The gradient of the wetland area is high to low from the southwest to the northeast. The distribution of positive (blue) and negative (red) breaks in shown in Panel B. The negative breaks appear to be more correlated to the location of the road but the distribution does not appear to be gradient specific.
Figure 3.10. A violin-plot of the up-gradient (left) and down-gradient (right) BFAST breaks. The sample size (number of pixels) is provided in the center of each violin plot. The vertical length of each violin plot represents the range of values within each year. The width of the plot shows the density of observations. The distribution of the magnitude of breaks were very similar (in terms of mean and density of observations) in up and down gradient areas. The mean of both up and down gradient sites is below zero (indicated by red dashed line).
DISCUSSION

Despite being one of the earliest and most prevalent anthropogenic disturbances on Earth’s surface, little is known about the impacts of roads in wetland areas. I used various RS analytical methods to identify roads and test the impacts of roads in remote wetland/peatland complexes in northern Alberta, Canada. Several key findings include:

- Despite the small relative size of roads on the landscape, the abrupt pixel change associated with road building is consistently identified using well-established CVA change detection methods.
- Objective 1: Changes in TC wetness and brightness after road construction were detectable at certain time intervals: immediate changes (less than one year after) were harder to detect than changes at three and five years after road building.
- Objective 2: When “wetter” wetland areas were intersected by roads, vegetation indices indicated they became drier. Yet, changes were not detected in “drier” wetlands more closely resembling upland forest.
- Objective 3: Analysis of a single wetland confirmed that the magnitude of change was largest directly adjacent to the road yet changes further from the road were more variable.

Surprisingly, the gradient did not affect the trajectory of wetland change (increasing wetness vs. dessication).

Next, I discuss these key findings in greater depth, along with limitations, significance to wetland RS monitoring in northern areas, as well as directions for future research.
Objective 1: RS trends did not return to pre-disturbance conditions after 5 years of monitoring

Wetlands did not recover to pre-disturbance conditions within 5-years after road-building (as would be expected under the time-lag hypothesis). However, a longer temporal period would likely need to be tested to confirm the regime-shift hypothesis (Figure 3.1). The lack of detectable wetland response to roads after 1-year may be due to several reasons. For consistency across multiple wetlands, the first year of assessment corresponded to the year the road was first visible. However, the annual time-step of imagery resulted in some uncertainty. For example, a road built in either September 1995 or July 1996 would both have 1996 as the first assessment year. This discrepancy may have contributed to the large range of values observed within the first year therefore resulting in additional time-steps to observe a consistent response to road building. In addition, the 1-year models, based upon one data point before and after construction, also limited the statistical power of 1-year BACI models.

Both theoretical frameworks of peatland restoration / recovery as well as practical applications support the regime shift and time-lag recovery hypotheses. Several ecological restoration theories and empirical results suggest that returning to pre-disturbance conditions requires intervention to re-establish feedbacks (Suding & Hobbs 2009; Martin & Kirkman 2009). Without interventions, some ecosystem thresholds are more likely to be surpassed, resulting in regime shifts (Suding & Hobbs 2009). However, peatland ecosystems are considered highly connected, complex, and adaptive ecohydrological systems (Belyea & Baird 2006; Dise 2009; Waddington et al. 2015). Peatlands rise and fall to adapt to rainfall and drought, maintain carbon balance during disturbances, and are able to respond to water-table drawdown (Bridgham et al. 2008; Dise 2009). Increasing the temporal range of this study would be required to confirm either hypothesis but it is clear from my findings (as well as others) that the impact of roads on
peatland trajectories is detectable on a short-term (5-year) basis. As best management practices for road building improve (Graf 2009) (e.g., culverts and permeable roadbeds), it is likely that the effects of roads will decrease in severity and thus improve recovery times. Therefore, it would be interesting to compare the regime shift vs. time-lag recovery hypothesis in older vs. more recently constructed roads.

*Trends in TC brightness and wetness indicated disrupted hydrological connectivity after road building*

Decreasing TC wetness (likely correlated with water table drawdown [Ordoyne & Friedl 2008]) was observed in down-gradient wetland areas supporting the expectation that road-building disrupts lateral and/or external hydrological connectivity. Likewise, increases in TC brightness have also been linked to decreasing wetland inundation (Huang et al. 2014; Li et al. 2015) and correlations between TC brightness and wetness has been previously found to be very high (r = -0.87 in Ordoyne & Friedl 2008). TC greenness is an effective vegetation index for northern and wetland environments due to its separate treatment of wetness, which often hinders the use of Normalized Difference Vegetation Index (NDVI) in such regions (Raynolds & Walker 2016). Changes in TC greenness, although not detected in this study, have been linked to boreal vegetation transitions following water table drawdown from bryophytes (lower TC greenness) to vascular shrubs and/or trees (higher TC greenness) (Raynolds & Walker 2016; Laiho et al. 2003). The absence of trends in TC greenness in my analysis may be due to a time delay of the effects of hydrologic control of wetland vegetation (Mitch and Gosselink 2007; Laiho et al. 2003) or the TC wetness change may be lower than the threshold for vegetation change. Vegetation trends may require monitoring over a longer time period due to lower soil temperatures, short growing seasons, the gradual nature of succession, and/or be site specific.
Water table drawdown in down-gradient wetland areas after road disturbance is corroborated by several field-based experiments in prairie, forested, and boreal areas (Jeglum 1975; Kahklen & Moll 1999; Patterson & Cooper 2007; Miller et al. 2015). Yet other experiments on the effects of water table drawdown on changes in vegetation found that the expected transition from bryophytes to shrubs and trees expected in drier, non-saturated conditions (Weltzin et al. 2003) took approximately 20 years to occur (Jukaine et al. 1995; Laiho et al. 2003). Studies such as these further provide some explanation for the lack of observed trends in TC greenness within a 5-year time span.

Although TC indices have been linked to wetland attributes in many previous studies (Ordoyne & Friedl 2008), additional field work would be helpful to verify correlations between RS and ground measurements (e.g. correlating a 10% decrease in TC wetness to a 10 cm decrease in water table level). The use of RS proxy indicators for water table depth (and other wetland attributes) are limited due to the variability of groundwater dynamics at a sub-pixel level and due to inaccuracy caused by obstruction of surface vegetation (Rahman et al. 2017). Moving forward, connecting RS indices with wetland attributes in different locations over broad spatial extents could help better enable utilization of RS products such as Landsat for wetland monitoring.

**Objective 2: Changes in the “wetter” wetlands**

The multi-wetland analysis indicated that RS trends in “wetter” wetlands were not necessarily observed in drier wetlands. The wetlands categories: HWLG, MWLG, and LWHG spanned the spectrum from open water to upland forest, when compared to the results of the 1985 baseline classification. My findings indicate that wetlands with higher initial TC wetness
are more susceptible to road disturbance than wetlands with lower TC wetness which likely had lower water table levels prior to disturbance.

Little research has compared the impacts of road-building among different wetland types. Based upon hypothesized hydrologic interactions (Willier 2017) relevant to peatland categorizations in northern Alberta’s boreal, I would expect groundwater-driven fens to be more susceptible to disruptions in hydrological connectivity than ombrotrophic bogs supported by precipitation. Also, due to the timing of my imagery (late growing season August), I expected water levels in bogs to be lower than in fens due to the lack of recent precipitation (Strack et al. 2007), perhaps fostering conditions most similar to drier LWHG-type wetlands throughout the region.

Overall, BACI designs are useful for detecting large, permanent changes in impacted sites compared to controls with several important limitations. While BACI results experiment provide statistically significant evidence of change, they do not indicate the biological significance or cause of change (Schwarz 2005, Underwood 1992). Further, the approach is not designed to identify transitional states but rather treats the “after” condition as a steady end state. As such, this assumption works for identifying permanent regime shifts but is less ideal for transitional states or dynamic recoveries. Lastly, BACI results are limited by the rigour in the selection of control wetlands; however, I implemented several measures to ensure control wetlands were undisturbed with similar initial conditions (TC indices) as impact sites.

**Objective 3: The magnitude of change was higher directly adjacent to roads**

The single wetland analysis provided greater spatio-temporal insight into the impacts of roads. Clustering of high magnitude impacts near the road disturbance was also observed in Bocking et al. (2017), where 100% tree mortality occurred on the up-gradient slope due to
flooding within a short distance (<83.5 cm) of the road. However, others found the effects of
roads on wetlands extended up to 500 m (Young et al. 2017). Further from roads, declines in TC
wetness were consistently less severe at both 4 to 10 years after disturbance. Negative breaks in
TC wetness may be more indicative of long-term climatic trends than road building as similar
magnitude trends were seen distributed broadly throughout the area. Use of controls (as in the
multi-wetland analysis) could help differentiate road impacts from future climatic changes. And
while it is possible that similar changes also occurred near roads in the multi-wetland study, such
complex spatial information was not easy to incorporate into a BACI design.

The majority of BFAST structural breaks (indicative of changing trends) occurred within
three years of road disturbance, matching the BACI results. Interestingly, trends in the second
and third year (1994 and 1995) were primarily positive (increasing wetness) which was inverse
to drying trends observed in other years and in the multi-wetland analysis. To provide insight
into these discrepancies, I assessed historical rainfall data from Fort McMurray, Alberta from
1985 to 2002 (Figure 3.11). There was higher than average precipitation at the time of image
acquisition in 1995, potentially resulting in increased TC wetness. However, increased TC
wetness trends were also observed in 1994 despite lower than average monthly precipitation in
August (Fig. 3.11), which does not fit the idea that increasing TC wetness is due to increased
summer rainfall (as was seen in 1995). Trends in TC wetness returned to primarily negative (i.e.
drying) after 1995 until the end of the study. Further investigation is required to confirm the
highlight one of the limitations of the BFAST method as it is not possible to attribute the cause
of the detected break.
Figure 3.11 Average monthly precipitation in Fort McMurray from 1985 to 2012. The red lines indicate 1994 and 1995, the years of higher increases in wetness (both years indicated on the figure). The black lines represent the remainder of the data from that period for reference. The figure demonstrates that in 1995, there was greater average precipitation in August (month 8). In contrast, the average precipitation in August of 1995 appeared lower than normal.
**Up- vs. down-gradient impacts of roads were not distinguishable**

In contrast to my own expectations and findings from previous research (Mader 2014; Miller et al. 2015; Bocking et al. 2017; Willier 2017), differences between up and down-gradient sides of a road were not detectable in the single wetland analysis. It is possible that the gradient was too shallow or that hydrological connectivity was maintained despite road disturbance. Further information on road substrate material, presence of culverts, sub-surface flow direction, and wetland type are needed to explain the absence of directional results at this wetland.

The limitations of using BFAST, like most RS change detection methods, include the inability to interpret the source of disturbance and to associate the magnitude of breakpoints with definitive ground processes (Verbesselt et al. 2011). Additionally, the use of an annual time-step results in fewer data points compared to incorporating intra-annual, seasonal data. The nature of pixel-based assessment (relevant to BACI and BFAST analysis) is subject to the errors inherent in RS data. RS data is subject to various types of measurement error, such as atmospheric effects, clouds, and geometric misalignments (Kennedy et al. 2014). There are additional limitations specific to the BAP that are discussed further below. Despite the design and study-specific drawbacks, the ability to set disturbance date and to view the trend results spatiotemporally was useful as an investigative tool in support of more robust analysis.

**Limitations of the BAP dataset**

The BAP Landsat database used in this study has several important limitations. For my study, the most important limitation is the treatment of missing pixels. Each pixel used in the annual composite image was scored based upon a series of criteria (sensor type, acquisition date, clouds and shadows, atmospheric opacity). The pixel with the highest sum score was used in the annual composite image. Pixels with unsuitable scores were treated as data gaps and filled in
using a time series based change mapping method called Composite2Change (C2C) (Hermosilla 2015a and 2016). The C2C process filled in data gaps with proxy (estimated) surface reflectance values using a piecewise linear interpolation of spectral values for individual pixel time series (Hermosilla et al. 2016). From 1984 to 1998 (when only TM images were available), 14.2% of pixels, on average, were filled in using proxy values. The combined use of TM and ETM+ after 1998, reduced data gaps to 5.9%. After the decommissioning of TM in 2011, the number of images was substantially less, resulting in an increase to 21.3% proxy values in the 2012 composite (Hermosilla et al. 2016). Note, the proportions of proxy values were calculated for all of Canada and are different for my study area. Interior areas with flat topography and less persistent cloud cover, such as my study area, have better data coverage than northern, coastal and mountainous areas. It is possible that the presence of proxy values, in isolated years and locations, distorted a small number of TC values extracted for statistical analysis.

**Implications for wetland monitoring**

Based upon these results and in consideration of improving EIA (Chapter 2) and natural resource management, several strengths and areas for improvement within my approach were identified. One strength included the use of GIS and RS indices to select control sites in a way that was effective as well as easy to document and is repeatable by others. My approach enabled selection of specific attributes for control sites (i.e. TC indices, distance to impact wetland, and corresponding wetland size) and was not limited to accessible, nearby sites. Insights from BACI also highlight its utility to evaluate the timeline of recovery or detect regime shifts. Coupled with the ability to select appropriate controls, the use of RS indices for monitoring recovery is an effective and fast way to confirm trend changes (or lack of change). Finally, the combination of multiple wetland and single wetland analysis provided unique insight into spatio-temporal
patterns across the landscape. The combination of these two methods provided an investigative tool to check broad multi-site trends, an essential part of scaling-up landscape analysis. Areas for improvement and further research are elaborated on below.

**Future Research**

To advance the results found in this analysis, future analysis is recommended within this body of work as well as broadly within the field (discussed in following Chapter IV Conclusions). An accuracy assessment of the main wetland types could use historical information (such as 1985 aerial photos) in combination with outlier detection techniques (as outlined in Nielson et al. 2008) that incorporate contemporary wetland information (Ducks Unlimited Canadian Wetland Inventory). These additional steps could help further validate the accuracy assessment using contemporary data provided in Appendix A.

Following the work of Harris et al. (2005, 2006, 2008) it would be interesting to further isolate the spectral signature for *Sphagnum* moss species using Landsat as an additional proxy for near-surface hydrology in peatlands. *Sphagnum* dynamics provide insight into changing water table levels, as well as shifting vegetation communities. Similarly, future work could aim to isolate the spectral signature of other common mosses or trees found in bogs (e.g. liverworts) and fens. Incorporation of several other RS indices, such as the Moisture Stress Index (MSI), the MODIS evaporation product and/or Landsat-derived evapotranspiration would also provide additional insights. Additional research should aim to expand into areas of discontinuous and continuous permafrost (further into northern Canada) and compare among other types of linear disturbances (i.e. seismic lines, pipeline right-of-ways).

Future research could also include making more explicit links between ecological mechanisms and RS break detection outcomes. Fieldwork is required to make these links, but is
often not possible in such large areas. Continuing collaboration between industry, field scientists, and RS specialists can ensure resources are used most efficiently. For example, as many industrial developments rely on fly-in access, airborne remote sensing collected in parallel may provide an opportune synergy. As the human footprint of development continues to expand into remote areas of northern Canada, it is increasingly important to have RS technologies and methods in place to monitor changes in terrestrial and aquatic ecosystems.
IV. CONCLUSION

Contending with pressing environmental issues in remote, northern areas requires acquiring data and implementing methods for understudied systems over broad spatial and temporal extents (Soranno et al. 2015). Combining smaller site-level assessment using aerial photos (as in Chapter 2) with imagery from satellite remote sensing (as in Chapter 3) provides an important way forward. While the need to “scale-up” environmental information has been recognized for decades, in practice, the use of broad-scale datasets for natural resource management remains limited by disparate, specialized, and often confidential approaches to data collection and reporting (Soranno et al. 2015). To address these needs within an EIA context, I explored two landscapes in northern Canada: a conceptual example in B.C. (Chapter 2) and an applied example in Alberta (Chapter 3).

In my second chapter, I proposed three main opportunities for integrating LE and ecosystem services (ES) into Environmental Impact Assessment (EIA) practice. With a focus on the Canadian EIA scoping process, I explored hypothetical, yet realistic, situations to demonstrate how the integration of scientific principles with open source data and software could improve EIA. First, I demonstrated how changing the extents of EIA spatial boundaries, through exclusion or inclusion of surrounding natural resource development projects, can potentially misrepresent cumulative impacts. Second, I used network analysis to show how even seemingly small, localized developments can disrupt regional connectivity. I also revealed tools for characterizing baseline vegetation using historical aerial photos to better convey effects of past stressors and improve the efficacy of restoration planning. Finally, I reviewed how a regulating ecosystem service framework can better account for water filtration services provided by
wetlands. All three of these approaches are straightforward to implement using publicly available data, thus provide sensible opportunities to improve EIA and concomitant decision-making.

Building on the observed changes likely due to flooding adjacent to a forestry road seen Chapter 2, in my third chapter I used RS indices to characterize road impacts to peatlands in northern Alberta. My work provided multi-scale insights by assessing RS spatio-temporal trends at two scales via multi- and single wetland analyses. To measure trends across multiple wetlands, I applied a common ecological approach, a BACI experimental design, to measure changes in RS indices before and after road building across multiple wetlands to test for disruption of hydrological connectivity. I then compared the trends observed from multiple-wetland analysis to pixel-based spatio-temporal trends at a single wetland using a BFAST monitor approach.

BACI was useful to detect change across multiple wetlands and to monitor the recovery (or lack of recovery) of wetlands. Pixel-based trends at a single site demonstrated that proximity to disturbance is important, as the majority of high magnitude changes were observed adjacent to road disturbances. Comparison of the two approaches provided a more complete picture of road-building disturbances and highlighted the importance of using long-term controls in RS analysis. My promising findings also support the need for continued research into associating ecological mechanisms with changes in RS indices. Next, I discuss some of the common threads found between the two chapters, several insights into EIA and environmental monitoring, and highlight some future research directions.

One of the persistent challenges in EIA is collecting sufficient data to match the temporal scale of impacts. To predict and later mitigate the impacts of a project/development, the temporal extent should include historical and contemporary pre-site baseline information and then account for a period of monitoring and restoration afterwards. However, this can be a large undertaking
and the temporal scale of a project is limited by the availability of site-specific historical data. The use of air photos and RS has the potential to fill in these gaps in different ways. Insights from Chapter 2 and 3 highlighted these. First in Chapter 2, historical air photos were used to characterize, quantify and map pre-disturbance conditions in riparian areas to set more appropriate targets for restoration. Historical aerial photographs were also used to demonstrate up-gradient flooding of water after road building that persisted on the landscape. To elaborate on the effects of roads on wetland areas in Chapter 3, RS data, specifically Landsat, was used to identify change in hydrological connectivity after disturbance compare to pre-disturbance and control wetland conditions. A combination of RS and BACI analysis also provided an effective method to monitor the timeline for recovery.

Another common thread between chapters is the need to explicitly account for roads in EIA and natural resource monitoring and management. Roads have many ecological impacts on habitat loss/fragmentation, aquatic and terrestrial biodiversity, spread of invasive species, sedimentation, increased particulate matter, and disrupted hydrological connectivity in wetlands (Chapters 2 and 3; Trombulak & Frissell 2000; Eigenbrod et al. 2009). Historically, roads were considered an inevitable and acceptable part of human-influenced landscapes. However, empirical evidence and observations have led to increasing concern over the many ecological impacts of roads, and now new road developments require EIAs in many countries (Karlson et al. 2014). Despite inclusion in the process, EIAs of road projects are generally considered poor due to neglect of cumulative effects (the landscape perspective) and uncertainties about their ecological effects in the scientific literature (van der Ree et al. 2015; Jaeger 2015). Continued research into the effects of roads and careful consideration of road placement in remote areas
needs to remain a priority for local governments, forestry companies, and natural resource managers.

The culmination of these two chapters highlights the importance of free and open data and software and the widespread utility of ‘ready-to-go’ RS databases such as the Canada-wide BAP Landsat database (White & Wulder 2014; Hermosilla et al. 2016). The increasing use of meta-analysis, shared data/software, implementation of landscape ecology theory and practice, and utilization of RS products is disproportionately important and timely in remote, vast, and/or northern areas. Finally, application of open source methods at different spatial scales (i.e. multiple wetlands vs. a single wetland) provides unique, and sometimes contrasting, insight into environmental impacts.

**Future Directions**

An important direction for future research is the development of RS environmental indicators for use in wetland EIA and monitoring site recovery. Due to the temporal and spatial requirements of EIA (Chapter 2), the dynamic inter- and intra-annual nature of wetlands, and the remote location of many natural resource extraction projects, there is great opportunity for the development of RS methods to be used in EIA. An example of RS application within EIA related to my thesis findings could be a requirement for project proponents to characterize baseline wetland conditions and to provide monitoring of site recovery (return or deviation from baseline conditions) during and after development. Monitoring of wetland restoration is perceived as a time-consuming and expensive task, often with unclear objectives (Reif & Theel 2017). However, to ensure effective restoration, it is critical that a monitoring system is established prior to project development and well-beyond the early stages of the project to track progress and intervene as required (Klemas 2013). Field programs are important and required in these
scenarios but are difficult to continue in perpetuity. Therefore, the use of RS could have important utility in restoration. The growth of RS use in restoration monitoring has increased over the past decade due to decreasing costs and improved technology (Klemas 2013). Further research is required to determine feasible use of RS in land and wetland restoration and in other aspects of EIA. Associated with this research should be a practical and specific application to relevant policy that dictates EIA outcomes.

Despite the proliferation of geospatial tools and approaches, use of decision-support tools to evaluate environmental policy has received limited academic interest (De Leeuw et al. 2010; Mayer & Lopez 2011). While the potential for geospatial researchers to participate in policy is lauded (Chapter 2 and 3; Mayer & Lopez 2011), their participation is often not linked to policy specifics (De Leeuw et al. 2010). Despite the successful implementation of RS into policy making (such as satellite detection of the hole in the ozone-layer culminating into a ban on chlorofluorocarbons), RS studies are seldom designed to answer specific policy outcomes. In the past, barriers to the use of landscape and RS analysis included cost, specialized skill, lack of data and software, inconsistent methods, and fluctuating political reasons (Mayer & Lopez 2011). As these barriers continue to crumble, future landscape GIS/RS research should focus on making more direct links to policy development, policy implementation, and evaluation to utilize the full potential of this expanding technology.
REFERENCES


Sutherland, IJ, Villamagna, AM, Ouellet Daillaire, C, Bennett, E, Chin, ATM, Yeung, ACY, Lamothe, KA, Tomscha, SA, Cormier, R. 2017. Undervalued and under pressure: A plea for greater attention toward regulating ecosystem services. Ecological Indicators (In Press).


APPENDIX A: Map Comparison

Ground verification of historical maps is notoriously challenging, if not impossible. Thus, I used a novel and creative approach to conduct a map comparison, which provides helpful context for the rigor of the 1985 baseline wetland map. The baseline wetland types (HWLG, MWLG and LWHG) were assessed by applying the same clustering algorithm used in 1985 to 2012 BAP imagery to compare with a contemporary wetland map. Due to differences in climatic conditions between 1985 and 2012, extrapolating the classification to a contemporary image did not identify the same individual wetland areas. Instead, this contemporary classification did identify wetlands with similar RS indices which are indicative of general wetland characteristics (i.e. hydrologic and vegetation features). I then compared the HWLG, MWLG, and LWHG wetland types in the contemporary classification to the province-wide Merged Wetland Inventory (MWI) produced by Alberta Environment and Parks ([AEP] 2018). Details about the MWI, results of the map comparison, and a brief discussion of the limitations of this analysis are provided next.

The MWI combined inventories from various geographic locations from 1998 to 2015, which were produced using three systems of wetland classification (Table A-1) to provide nearly provincial-wide mapping coverage. While these three classifications each identified distinctly different classes, their classes could ultimately be amalgamated into five wetland types of the Canadian Wetland Classification System (CWCS): bog, fen, marsh, swamp, and shallow water. While the resolution differed by region and data availability, the minimum spatial resolution is 30 m (with finer-scale information at some locations). The results of field verification, despite being conducted in some limited portions of the MWI, were not available for the merged product (AEP 2018). A limited accuracy assessment of wetland and non-wetland classes was completed within my study region (Hird et al. 2017) yielding an overall accuracy of 85.4% with a Kappa coefficient of 0.672. The per class accuracies for detailed classes of the MWI remains unclear.

Table A-1. Description of the three wetland classification systems used in combination to create the Merged Wetland Inventory (MWI) by Alberta Environment and Parks (AEP) circa March 2017.

<table>
<thead>
<tr>
<th>Wetland Classification System</th>
<th>Source of Imagery</th>
<th>Resolution or Minimum mapping unit (MMU)</th>
<th>Original Classes</th>
</tr>
</thead>
<tbody>
<tr>
<td>Ducks Unlimited Canada Boreal Enhanced Wetland Classification System (EWC)</td>
<td>Landsat 5, 7 ETM+, and 8 OLI</td>
<td>30 m</td>
<td>19</td>
</tr>
<tr>
<td>SPOT Grassland Vegetation Inventory (GVI) Lentic Classification</td>
<td>SPOT 4 and 5 MSS</td>
<td>10 m</td>
<td>32</td>
</tr>
<tr>
<td>High Resolution: Stereo models (using Canadian Wetland Inventory standards)</td>
<td>Aerial photography(various scales)</td>
<td>0.02 – 0.1 ha (200 - 1000 m²)</td>
<td>5</td>
</tr>
</tbody>
</table>

The accuracy of my contemporary classification was assessed at 400 stratified random points whereby 100 were distributed in each of three wetland types (HWLG, MWLG, and LWHG) as well as in non-wetland areas. These classes were then compared with corresponding
MWI classes using ArcGIS (version 10.5). HWLG and MWLG classes were merged into one combined LG class based on Chapter 3 where I detected a high level of similarity in their defining characteristics. More specifically, given their low TC greenness in both classes, LG wetlands were reasonably assumed to consist of treed peatlands with high water tables. The LG class therefore most closely aligned with the swamp class in MWI. Swamps are further defined by the CWCS as having either peatland or mineral soils with the water table at (or near) the surface, along with coniferous/deciduous trees or tall shrub cover (AEP 2018). In contrast, higher TC greenness (as in LWHG) was assumed to reflect shrubs/graminoids communities commonly found in bogs and fens. As such, for this map comparison, LWHG was evaluated relative to a combined “bog and fen” MWI class.

The comparison between my contemporary classification and the Alberta MWI yielded an overall accuracy of 63% with a Kappa coefficient of 0.44. User’s accuracy for the LG, LWHG, and non-wetland areas varied from 50%, 80%, to 71%, respectively. While the producer’s accuracy ranged from 77% (HWLG/MWLG), to 43% (LWHG), to 84% for non-classified areas.

Table A-2. Results of a comparative accuracy assessment between my 2012 BAP map and MWI. Green entries indicate the suspected wetland MWI class that matched with the 2012 classification.

<table>
<thead>
<tr>
<th>2012 Classification</th>
<th>Swamp</th>
<th>Bog</th>
<th>Fen</th>
<th>Other*</th>
<th>Row totals</th>
<th>Commission Errors</th>
<th>User's Accuracy (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td>LG (HW &amp; MW)</td>
<td>99</td>
<td>25</td>
<td>63</td>
<td>13</td>
<td>200</td>
<td>101</td>
<td>50%</td>
</tr>
<tr>
<td>LWHG</td>
<td>19</td>
<td>27</td>
<td>53</td>
<td>1</td>
<td>100</td>
<td>20</td>
<td>80%</td>
</tr>
<tr>
<td>Not classified</td>
<td>10</td>
<td>8</td>
<td>11</td>
<td>71</td>
<td>100</td>
<td>29</td>
<td>71%</td>
</tr>
<tr>
<td>Column totals</td>
<td>128</td>
<td>60</td>
<td>127</td>
<td>85</td>
<td>400</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Omission Errors</td>
<td>29</td>
<td>107</td>
<td>14</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Producer's Accuracy</td>
<td>77%</td>
<td>43%</td>
<td>84%</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

Note: * refers to areas identified as open water and marsh, which were not included in the baseline assessment.

The accuracy assessment results confirmed that the baseline wetland classes spanned the spectrum from open water to upland forest and that the classes represent the relative hydrodynamics of the wetland areas at the time of image acquisition and less so a consistent wetland type. Importantly, the results confirm the similarity of the HWLG and MWLG classes and support the inclusion of these two classes in future analysis. Due to the dynamic nature of TC wetness, the results highlight the importance of TC greenness as a stable, long-term indicator of wetland response / recovery to disturbances and changes in hydrologic condition. This finding corroborates the results of my BACI analysis and reiterates my recommendation to extend the monitoring timeline of vegetation indices.

There are several important limitations in both datasets. Within my classification, it is possible that the original TC clustering used in 1985 is not representative of the climatic
conditions in 2012 and characteristics of wetland types have changed. It is possible that an unsupervised classification using TC values would have different results if completed solely on the 2012 BAP composite. In the MWI dataset, details of the procedures and results form the field-level ground verification are not readily available however the metadata states that geoprocessing errors are likely present in the data due to the amalgamation of several large datasets. The MWI has inherent, internal consistencies, and data gaps that limit its practical use for small-scale wetland identification (Hird et al. 2017). The limitations of the contemporary BAP wetland classes and the MWI data mean that this accuracy assessment is useful as a preliminary tool but is not expected to replace the use of air photos and other accuracy assessment methods as stated in Chapter 3 – Future Research.
APPENDIX B
Alberta Biodiversity Monitoring Institute (ABMI) Human Footprint Index (HFI) Metadata

ABMI Human Footprint Inventory for 2012 conditions (Version 3) from the Alberta Biodiversity Monitoring Institute was used, in whole or part, to create this product. More information on the Institute can be found at: http://www.abmi.ca.

<table>
<thead>
<tr>
<th>HFI Layer</th>
<th>Data Source</th>
</tr>
</thead>
<tbody>
<tr>
<td>Roads</td>
<td>Alberta Environment and Parks (AEP) Base Layer database</td>
</tr>
<tr>
<td>Reservoirs</td>
<td>AEP</td>
</tr>
<tr>
<td>Borrow-pits, sumps, dugouts</td>
<td>AEP</td>
</tr>
<tr>
<td>Rail-lines hard surface</td>
<td>AEP</td>
</tr>
<tr>
<td>Canals</td>
<td>AEP</td>
</tr>
<tr>
<td>Vegetated roads, trails, railways</td>
<td>AEP</td>
</tr>
<tr>
<td>Mine sites</td>
<td>AEP with additions from ABMI using 2012 SPOT image</td>
</tr>
<tr>
<td>Industrial sites</td>
<td>AEP with additions from ABMI using 2012 SPOT image</td>
</tr>
<tr>
<td>Active Well Sites (Energy)</td>
<td>AEP with additions from ABMI using 2012 SPOT image</td>
</tr>
<tr>
<td>Landfill</td>
<td>Created by ABMI with SPOT imagery</td>
</tr>
<tr>
<td>Other vegetated sites and recreation</td>
<td>Created by ABMI with SPOT imagery</td>
</tr>
<tr>
<td>Transmission lines</td>
<td>AEP</td>
</tr>
<tr>
<td>High density livestock</td>
<td>Created by ABMI with SPOT imagery</td>
</tr>
<tr>
<td>Urban and rural residential</td>
<td>Created by ABMI with SPOT imagery</td>
</tr>
<tr>
<td>Well sites – Abandoned</td>
<td>AEP with additions from ABMI using 2012 SPOT image</td>
</tr>
<tr>
<td>Cultivation</td>
<td>Created by ABMI with SPOT imagery</td>
</tr>
<tr>
<td>Cut blocks</td>
<td>AEP, Private forestry companies, with additions from ABMI using SPOT</td>
</tr>
<tr>
<td>Pipelines</td>
<td>AEP</td>
</tr>
<tr>
<td>Seismic lines</td>
<td>AEP</td>
</tr>
<tr>
<td>Disturbed vegetation</td>
<td>Created by ABMI using 2012 SPOT imagery</td>
</tr>
</tbody>
</table>