Characterizations of boreal anthropogenic disturbance regimes from multi-scalar Earth observations

by

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Abstract

Anthropogenic disturbance regimes are anticipated to overwhelm Earth's ecosystems during the Anthropocene. Boreal forests are particularly at risk of significant transition due to human appropriation of renewable and non-renewable resources. Forestry and energy development in the boreal forest have three primary ecological consequences: suppression of historical disturbance regimes such as fire; emergence of novel ecosystems; and the eradication of ecological memory, which maintains ecological integrity. The objective of this dissertation is to improve our understanding of the pattern characteristics of anthropogenic disturbance regimes in order to mitigate the negative, unintended outcomes of managed boreal forests.

Anthropogenic disturbance from forest harvesting and energy development was mapped for industrialized landscapes of Alberta, Canada between 1949 and 2012. A comparative analysis using spatial models of unsuppressed fires sampled across Alberta and Saskatchewan and aerially-interpreted forest inventory data revealed that the anthropogenic disturbance patterns were beyond the historical range-of-variability in terms of disturbed area, largest patch size, and undisturbed forest remnants. When the spatial data were segmented based on a recent period of intensive energy development, it was determined that energy development in Alberta was a major driver of cumulative anthropogenic disturbance patterns. Levels of undisturbed forest remnants within anthropogenic disturbances declined between 18-34% and edge density increased between 15-175% following energy development.

Landscape-level patterns of forest cover changes were assessed using a time series of satellite imagery between 1985 and 2010. Forest disturbance was classified as resource extraction or fire in the Foothills of Alberta with 94% overall accuracy. The rate of resource extraction exceeded fire, accounting for 86% of annual forest disturbance, indicating that fire was suppressed in the landscape. A time series pattern analysis approach applied across Canada demonstrated that managed boreal forests were associated with rising edge density, declining core forest cover, and declining largest forest patch size. Boreal forests that had low disturbance rates were characterized by inherent forest cover pattern variation.

This dissertation advanced new perspectives on conceptualizing, detecting, and characterizing patterns of anthropogenic disturbance regimes. Future work is identified primarily around the development and interpretation of landscape structure thresholds and transition indicators.

Preface

The research questions and objectives of this dissertation were originally conceived from discussions between me and my supervisory committee. Portions of this dissertation appear as co-authored publications in peer-reviewed journals. For these publications, I performed the primary research, data analysis and interpretation, and prepared the final manuscript:

- Chapter 2: **Pickell, P.D.**, Gergel, S.E., Coops, N.C., and Andison, D.W. (2014). Monitoring forest change in landscapes under-going rapid energy development: Challenges and new perspectives. *Land* 3(3): 617-638. doi:10.3390/land3030617
- Chapter 3: **Pickell, P.D.**, Andison, D.W., and Coops, N.C. (2013). Characterizations of anthropogenic disturbance patterns in the mixedwood boreal forest of Alberta, Canada. *Forest Ecology and Management* 304: 243-253. doi:10.1016/j.foreco.2013.04.031
- Chapter 4: **Pickell, P.D.**, Andison, D.W., Coops, N.C., Gergel, S.E., and Marshall, P.L. (2015). The spatial patterns of anthropogenic disturbance in the western Canadian boreal forest following oil and gas development. *Canadian Journal of Forest Research* 45(6): 732-743. doi:10.1139/cjfr-2014-0546
- Chapter 5: **Pickell, P.D.**, Hermosilla, T., Coops, N.C., Masek, J.G., Franks, S., and Huang, C. (2014). Monitoring anthropogenic disturbance trends in an industrialized boreal forest with Landsat time series. *Remote Sensing Letters* 5(9): 783-792. doi:10.1080/2150704X.2014.967881
- Chapter 6: **Pickell, P.D.**, Hermosilla, T., Frazier, R., Coops, N.C., and Wulder, M.A. (2016). Forest recovery trends derived from Landsat time series for North American boreal forests. *International Journal of Remote Sensing* 37(1): 138-149. doi:10.1080/2150704X.2015.1126375
- Chapter 6: **Pickell, P.D.**, Coops, N.C., Gergel, S.E., Andison, D.W., and Marshall, P.L. Managed boreal forests drive rapidly changing landscape patterns over the last three decades in Canada. (in review)

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List of Abbreviations

%IR – Disturbance event percentage area of island remnants

%DP – Disturbance event percentage area of disturbance patch

%LDP – Disturbance event percentage area of largest disturbed patch

%MR – Disturbance event percentage area of matrix remnants

%TR – Disturbance event percentage area of total remnants

Al-Pac – Alberta-Pacific Forest Industries

ANC – Alberta Newsprint Company

AVI – Alberta Vegetation Inventory

B5 – Mid-infrared Landsat spectral band 5

BAP – Best available pixel

CDF – Cumulative distribution function

CRV – Current range of variability

DI – Disturbance Index

EA – Disturbance event area

EBM – Ecosystem-based management

ECDF – Empirical cumulative distribution function

ED – Edge density

FMA – Forest management area

GIS – Geographic Information System

HRV – Historical range of variability

HWP – Hinton Wood Products

LDPA – Disturbance event largest disturbed patch area

LiDAR – Light Detection and Ranging

LIRA – Disturbance event largest island remnant area

LPI – Largest patch index

MDPA – Disturbance event mean disturbed patch area

MIRA – Disturbance event mean island remnant area

MMU – Minimum mapping unit

MPE – Multiple-patch disturbance event

MWW – Mann-Whitney-Wilcoxon test

MODIS – Moderate Resolution Imaging Spectroradiometer

NBR – Normalized Burn Ratio

NDVI – Normalized Differenced Vegetation Index

NEPTUNE – Novel Emulation Pattern Tool for Understanding Natural Events

PD – Patch density

SPE – Single-patch disturbance event

TCG – Tasseled Cap Greenness

TCT – Tasseled Cap Transformation

VCF – Vegetation Continuous Field

VCT – Vegetation Change Tracker

WRS-2 – Worldwide Reference System 2

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Finally, I extend my gratitude and thanks to my family, friends, and lab mates at the Integrated Remote Sensing Studio for endorsing this expedition. My love, Jordan Baker Pickell, provided me with encouraging support throughout my studies and kept me critical of current paradigms and open to alternative possibilities.

To all my teachers.

Chapter 1

Introduction

1.1 Boreal forests as a case study during the Anthropocene

The impacts from human appropriation of Earth's systems are informally recognized by the characterization of a new global geologic epoch known as the Anthropocene (Crutzen and Stoermer, 2000). During the Anthropocene, human activities are expected to overwhelm the carrying capacity of Earth's systems (Rockström *et al.*, 2009) as well as the natural forces that historically shaped them (Steffen *et al.*, 2007). Globally, humans are estimated to appropriate 23% of renewable freshwater (Postel *et al.*, 1996) and approximately 24% of potential net primary production (Haberl *et al.*, 2007). In some cases, human appropriation of Earth's systems are beyond the planet's sustainable capacity (Rockström *et al.*, 2009).

The impacts of human activities on Earth's systems have been conceptualized in a number of ways. At the global level, Vitousek and colleagues (1997) recognized that human alteration of Earth's systems is so pervasive that in many cases we cannot fully understand those systems without also accounting for the dominance of the human signal. Ellis and Ramankutty (2008) extended this thinking with the conceptualization of anthropogenic biomes or *anthromes* across the planet. On a more practical level, humans have both direct and indirect impacts on ecosystem development. Direct anthropogenic impacts can be characterized as those that contribute to the development of desirable ecosystem patterns and structures (*e.g.*, the accumulation of biomass and wood fiber in forests; Odum, 1969) while indirect impacts are primarily driven by more complex social and economic factors of human cultural development (*e.g.*, climate change and urbanization; Ellis, 2015). I use the term *anthropogenic disturbance regime* in this dissertation to describe the emergence of ecological patterns and structures as a result of direct human appropriation of the terrestrial biosphere.

Boreal forests are the most common forest type on the planet, comprising some 30% of global forest cover (Brandt *et al.*, 2013). This biome provides the majority of the renewable freshwater supply in the Northern Hemisphere (Oki and Kanae, 2006) and sequesters between

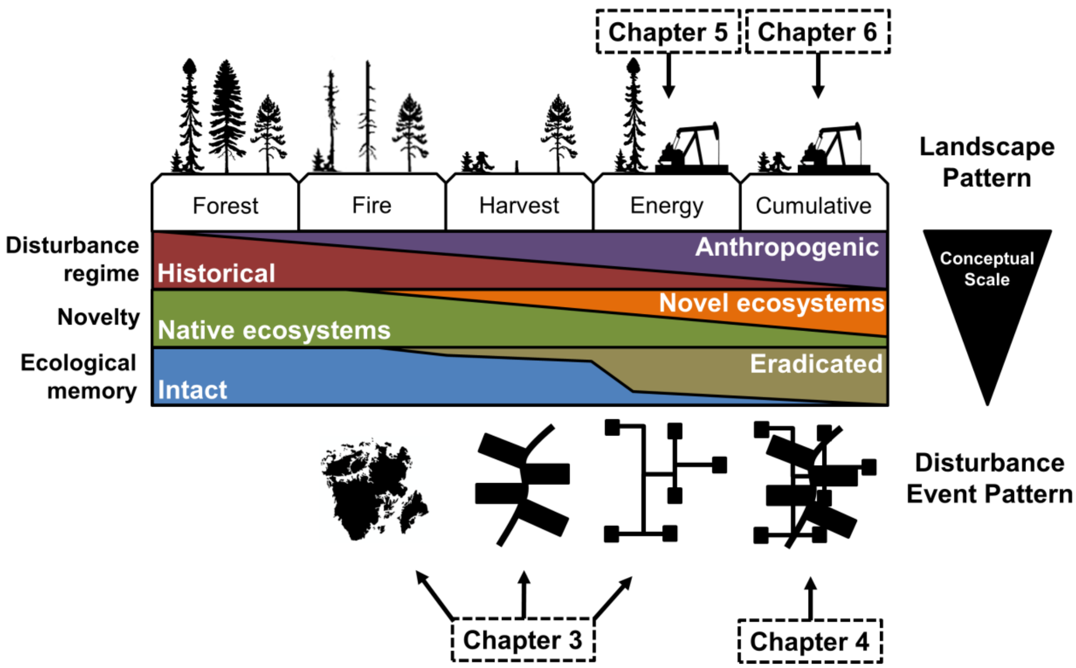
0.4 and 0.6 Pg of carbon per year (Pan *et al.*, 2011), while providing numerous other ecosystem goods and services (Shvidenko *et al.*, 2005; Statistics Canada, 2013). The biome is characterized by a low human population density (Ellis and Ramankutty, 2008) and high abundance of renewable and non-renewable resources. Moreover, the biome is considered largely intact, dominated by large tracts of continuous forest cover (Lee *et al.*, 2003; Potapov *et al.*, 2008).

The signature of the Anthropocene can be seen in boreal forests. So far in the 21st century, boreal forests have undergone the most significant changes in cover compared with other forest types (Hansen *et al.*, 2013). Globally, boreal forests are undergoing major transitions: unprecedented climate change (Gauthier *et al.*, 2015); disease and insect outbreak (McCullough *et al.*, 1998); wetland loss (Schindler, 2001); and invasion of non-native species (Sanderson *et al.*, 2012). Moreover, there is strong evidence to suggest that these changes were precipitated by human land and resource use (Foley *et al.*, 2013). All of this evidence should be interpreted in light of the fact that these changes have primarily occurred within the last two centuries in North America and this presents a special challenge for characterizing the anthropogenic disturbance regime (Ellis, 2015). On the other hand, the spatial and temporal transition between intact ecosystems and highly modified ecosystems can be demarcated quite easily in North American boreal forests due to the relatively recent industrialization of southern forests (Timoney, 2003). Moreover, a large portion of northern boreal forests are considered intact with relatively low industrial anthropogenic impacts (Lee *et al.*, 2003).

Human use of boreal forests has three primary ecological consequences: suppression of historical disturbance regimes such as fire; emergence of novel ecosystems; and the eradication of ecological memory, which is a key factor in maintaining ecological integrity (Figure 1.1).

Suppression of historical disturbance regimes. Approximately two-thirds of boreal forests are formally managed in the form of fire suppression efforts, protected areas, or tenured wood production (Gauthier *et al.*, 2015). The exclusion of fire from many Canadian boreal forests and the dominance of forest management in the recent century have resulted in the homogenization of forest stand age, increased forest fuel loadings, and the simplification of forest spatial patterns (Gauthier *et al.*, 2015).

Figure 1.1 Conceptual diagram developed in Chapter 2 depicting the three primary themes that relate the research chapters of this dissertation.



Emergence of novel ecosystems. Anthropogenic boreal forests have been linked with the emergence of novel ecosystems comprised of non-native species (MacFarlane, 2003) and changing trophic relations (Hebblewhite *et al.*, 2005; Laliberte and Ripple, 2004). There is some support for the hypothesis that biodiversity peaks at intermediate levels of anthropogenic disturbance, which indicates the emergence of novel ecosystems in managed boreal forests (Mayor *et al.*, 2012).

Eradication of ecological memory. Human activities in boreal forests can often result in degradation of seed and bud banks due to soil compaction by heavy equipment or recreation (Lee and Boutin, 2006). Eradication of ecological memory can compromise boreal forest resilience (Drever *et al.*, 2006) and impact the relative success of native vegetation recovery following perturbation (Paine *et al.*, 1998).

Recent concerns have emerged that forest management may not support resilient boreal forests (Drever *et al.*, 2006). Hypotheses have been raised that the spatial patterns of anthropogenic disturbance are critically important for maintaining boreal biodiversity (Hunter, 1993). Boreal forests are well-suited to so-called ecosystem-based management approaches that attempt to emulate historical landscape processes (Cumming, 1997) because large fires account for the majority of area burned in most boreal forested landscapes of Canada (Cumming, 2001a). A predominant belief is that forest management can and has replaced fire with harvesting in the landscape, at least within managed Canadian boreal forests (Johnson *et al.*, 1998). The dispersal of forest harvesting in space and time across boreal forested landscapes has been a primary strategy of forest management in order to mitigate the impacts of clear cutting (McRae *et al.*, 2001).

Evidence suggests that the dispersed harvesting strategy has had unintended ecological consequences at the landscape level, primarily related to increased forest edge and road networks (Franklin and Forman, 1987). Thus, new approaches have been proposed to harvest larger patches with more forest retention (*i.e.*, undisturbed forest) in order to maintain historical boreal forest intactness and ecological integrity (Carlson and Kurz, 2007; Delong and Tanner, 1996; Delong, 2002; Hunter, 1993). The evaluation of the relative success of these approaches compared with traditional dispersed harvesting requires a better, more descriptive measures useful for comparing all types of disturbance (Andison, 2012). Additionally, such an approach is needed to assess whether anthropogenic disturbance regimes are beyond an historical range-of-variability in terms of fire burn patterns (Landres *et al.*, 1999; Morgan *et al.*, 1994).

Anthropogenic disturbance regimes may be characterized at a range of spatial scales, and a multi-scalar approach is necessary to improve our understanding of patterns of anthropogenic appropriation of boreal forests. Patterns can be characterized at the scale of a single disturbance event (*e.g.*, fire burn patterns) and at broader regional scales that might represent a nominal fire regime based on burning patterns and rates (Andison and McCleary, 2014; Andison, 2012). Similarly, patterns may be characterized for landscapes, which may vary in scale and are defined by the relationship between extent and grain (Gergel, 2007). Thus, to improve our understanding of boreal anthropogenic disturbance regimes, it is necessary to evaluate pattern outcomes for anthropogenic disturbance events in multiple regions and pattern outcomes for landscapes that represent the range of conditions of boreal forests across North America.

Boreal forests have seen an incredible emergence of energy sprawl from non-renewable energy sources since 2000. New appropriations of energy sources in boreal forests contribute significantly to forest loss, fragmentation, and biodiversity declines (Butt *et al.*, 2013). In particular, the exploitation of hydrocarbon and mineral resources throughout Canadian boreal forests has significant consequences for cumulative land use impacts (Schneider *et al.*, 2003). The impacts of energy sprawl in boreal forests are poorly understood and are likely to be a critical area of research during the Anthropocene (McDonald *et al.*, 2009).

New understandings of the cumulative impacts of anthropogenic disturbance regimes are necessary to mitigate unintended ecological consequences. Determining how non-renewable energy sprawl has altered boreal anthropogenic disturbance regimes is a critical knowledge gap in our understanding of human appropriation of the biosphere. In the coming decades, energy demand is expected to increase (USEIA, 2013) with the expansion of non-renewable infrastructure (INGAA, 2014) and land use sprawl (McDonald *et al.*, 2009). New linkages are required that bridge spatial patterns at the disturbance event level with landscape level patterns of boreal forest cover in order to understand the cumulative impacts from appropriations by forest management and the energy sector.

New approaches for detecting and characterizing patterns of anthropogenic disturbance regimes are needed at the landscape level. The relatively low population density and vastness of boreal forests makes extensive ground campaigns unfeasible. Remote sensing technologies are best suited to monitoring large areas over time with high efficiently and reproducibility (Wulder *et al.*, 2008a). The Landsat satellite image archive represents the largest collection of freely available Earth observations that are appropriate for characterizing patterns at the landscape level (Woodcock *et al.*, 2008; Wulder *et al.*, 2008b). These data have been utilized to characterize forest cover changes across the planet (Hansen *et al.*, 2013). The recent changes and trends in landscape level patterns of the distribution and configuration of boreal forest cover remains a critical knowledge gap. Analysis of landscape level patterns trends over the satellite record could provide profound insights into how anthropogenic disturbance regimes have influenced pattern outcomes across boreal forests of Canada.

1.2 Critical research needs during the Anthropocene

The Anthropocene is positioned to be an era of incredible change for boreal forests and as such it necessarily requires new perspectives on how we conceptualize, detect, and characterize the changes. Therefore, I have identified the following research needs that will be addressed in this dissertation:

- A reconceptualization of disturbance during the Anthropocene
- A common and transferrable spatial language and models for delimiting disturbance
- New methods for detecting, mapping, and characterizing disturbance

1.3 Research questions

The following questions are posed in order to meet the research needs outlined in section 1.2:

1. How can anthropogenic disturbance be conceptualized as a disturbance regime?
2. Does forest harvesting emulate fire patterns?

3. What are the cumulative anthropogenic disturbance patterns before and after energy development?
4. How can anthropogenic disturbance be detected from satellite observations?
5. How have boreal forest spatial patterns changed over time?

1.4 Dissertation overview

Three themes relate all of the research carried out in this dissertation: suppression of historical disturbance regimes; emergence of novel ecosystems; and the eradication of ecological memory. These themes are reviewed further in Chapter 2 and presented here as a conceptual diagram of the flow of the dissertation in Figure 1.1.

Chapter 2 provides an overview of anthropogenic disturbance and the challenges associated with assessing impacts of energy development across North America. New perspectives of anthropogenic disturbance regimes emerge from a critical inquiry and engagement with the peer-reviewed literature.

Chapter 3 investigates anthropogenic disturbance event patterns of forest harvesting and energy development relative to fire. A novel spatial language is introduced and used to map disturbance events arising from resource extraction in Alberta, Canada. A comparative analysis was undertaken to determine whether anthropogenic disturbance patterns from forest harvesting and energy development are within the historical range-of-variability for a suite of spatial pattern indices.

Chapter 4 investigates the interaction of forest harvesting and energy development on cumulative anthropogenic disturbance event patterns following a major period of energy development in Alberta, Canada. A key outcome of this chapter was the temporal segmentation of the anthropogenic disturbance regime in order to assess the change in disturbance patterns.

Chapter 5 investigates the methods for detecting anthropogenic disturbance signals from satellite imagery in a highly industrialized landscape in Alberta, Canada. A spectral trend analysis was applied to a time series of Landsat satellite imagery from 1985 to 2010 in order to map forest disturbances. It was demonstrated that anthropogenic disturbance could be detected from satellite observations using a combination of linear discriminant analysis and spectral trend analysis.

Chapter 6 investigates the temporal changes to landscape-level patterns for boreal forested landscapes across Canada. Firstly, forest cover changes are mapped using a spectral trend analysis and forest recovery model. A suite of spatial landscape pattern indices were computed at annual time steps in order to evaluate how boreal forest spatial patterns have changed over time.

Chapter 7 concludes with the major findings, innovations, and limitations of the dissertation as well as provides some recommended directions for future research.

Chapter 2

Overview of anthropogenic disturbance regimes

2.1 Human transformation of forests during the Anthropocene

Major transformation of forests by anthropogenic causes remains a critical impetus for quantifying landscape structure in ecology. Forested ecosystems are important for their role in sequestering carbon from the atmosphere, regulating ecological functions, retaining the largest stock of biomass than any other land type, and providing ecosystem goods and services that contribute to economic activity and subsistence of forest-dwelling people (Shvidenko *et al.*, 2005). Globally, forests cover approximately 31% of the terrestrial surface (4.033 Bha) and 17% of global forest cover (678 Mha) occurs in North America (FAO, 2010). Forests worldwide have undergone significant changes in coverage, particularly in the southern hemisphere (FAO, 2010). In contrast, North American forest coverage has increased over the past 100 years (but see Aide *et al.* (2012) for recent net forest losses in Latin America and the Caribbean) and has remained relatively stable over recent decades (FAO, 2010). Despite the stability of overall forest cover in North America, substantive changes in the arrangement and fragmentation of forest cover has occurred driven by forest management and fire suppression, and increasingly from energy development (Barbier *et al.*, 2010).

Anthropogenic impacts to forests can be challenging to detect and characterize and the literature lacks a synthesis of anthropogenic disturbance regimes, especially for land use change related to energy development (Harden *et al.*, 2014; Marris *et al.*, 2013). This review discusses the challenges associated with defining and characterizing anthropogenic disturbance in forests and proposes a new perspective for conceptualizing anthropogenic disturbance regimes. I focus on ecological change driven by energy resource development. I synthesize the major impacts of anthropogenic disturbance in forests related to oil and gas extraction and review the major advancements for characterizing and detecting anthropogenic disturbance regimes using multi-scalar Earth observations.

2.2 Conceptualizing forest land use as a disturbance regime

Human land use activities since the Industrial Revolution have significantly altered all of the terrestrial biomes of the biosphere (Ellis, 2011; Ellis *et al.*, 2010; Foley *et al.*, 2013; Smith, 2007; Steffen *et al.*, 2007) and cast Earth into a new geological epoch informally recognized as the Anthropocene (Crutzen and Stoermer, 2000). The conception that humans disturb "natural" systems has become untenable because most biomes have been significantly transformed by novel anthropogenic processes into humanized biomes or anthromes (Ellis and Haff, 2009; Ellis and Ramankutty, 2008; Ellis *et al.*, 2010; Marris *et al.*, 2013). Energy development has been a driver of forest land use change during the Anthropocene, providing both the means (*e.g.*, gasoline-powered motors) and the ends (*e.g.*, mineral and energy resources) of landscape change, which has also been recognized as a major cause of catastrophic biodiversity loss worldwide (Butt *et al.*, 2013; McDaniel and Borton, 2002). Thus, conceptualizing forest land use as a disturbance regime will be central to any understanding of ecological patterns during the Anthropocene (Harden *et al.*, 2014). Conventional measures of patch-based disturbance patterns and disturbance regimes are not well-positioned to meet future research needs for investigating the spatially continuous and heterogeneous effects that are observed for anthropogenic disturbance to forested ecosystems. In the following subsections, a working definition of anthropogenic disturbance is proposed and the challenges associated with characterizing anthropogenic disturbance regimes are discussed.

2.2.1 Defining anthropogenic disturbance

I define anthropogenic disturbance as any change to ecosystem composition, structure or function caused by human resource utilization. Any definition of anthropogenic disturbance is necessarily relative to the scale at which the analysis or observation is applied. For the broadest scales of observation, forest canopy modification is usually sufficient to describe regional, continental and global trends of pattern and process (Masek *et al.*, 2013). However, a patch-based definition of anthropogenic disturbance will miss many of the finer scale impacts of human land uses that are often the most pervasive drivers of landscape structure.

Forest land management practices in North America over the last two decades have shifted toward "low impact" strategies that aim to attenuate the degree to which forest cover is modified, thereby limiting the ability to detect and characterize many anthropogenic disturbances using cover as a sole indicator (MacFarlane, 2003). The impacts from these finer scale disturbances are typically not discrete, and do not conform to the traditional conception of disturbance (White and Pickett, 1985). White and Pickett (1985) define disturbance at the ecosystem scale as "any relatively discrete event in space which alters the physical structure of the environment and disrupts the availability of resources." Anthropogenic disturbance of forests is often mapped with discrete perimeters where forest cover has been completely or partially removed, but many anthropogenic impacts can and often do exceed these boundaries (Dyer *et al.*, 2001). Discreteness refers to the ability to thematically distinguish levels of vegetation condition, which usually occurs as a gradient between the binary "disturbed" and "undisturbed" states. Not all forms of anthropogenic disturbance are discrete in either the spatial or temporal domain, leading to a range of heterogeneous effects for forested ecosystems that can be challenging to assess and characterize using conventional measures of disturbance regimes. As a result, it is especially challenging and controversial to delimit the spatial boundary of anthropogenic disturbance.

2.2.2 Challenges for characterizing anthropogenic disturbance regimes

Disturbance regimes are conventionally described using measures of ecological change for a given location and period of time. The most commonly quantified measures include disturbance extent, severity, persistence, intensity, frequency, and seasonality. However, characterizing forest land use using conventional measures of disturbance regimes is exceptionally challenging. For example, what is analogous to extent and frequency for a disturbance regime that expands, disturbs, and displaces in perpetuity (*e.g.*, exurban development and land conversion)? Many of the conventional measures of disturbance regimes do not adequately capture the impacts of anthropogenic disturbance, thus, new perspectives on anthropogenic disturbance regimes are necessary for understanding forest land use change during the Anthropocene.

Extent is an important measure of disturbance for assessing the direct ecological influence of a disturbance, but it is inadequate for capturing non-discrete, linear disturbances.

Using extent as an indicator of anthropogenic influence would severely underestimate the ecological impacts of anthropogenic disturbance and make it appear as though the influence is less or smaller than the change that the ecosystem actually undergoes. In such cases, gradient or surface metrics would be preferred to extent for assessing the influence of disturbance (McGarigal *et al.*, 2009), especially related to resource utilization in forests. Despite the limited availability of appropriate metrics to characterize extent, edge density is a standard metric used to estimate extent related to linear disturbances (*e.g.*, $km \cdot km^{-2}$) and is useful for making comparisons between landscapes or regions (Heilman *et al.*, 2002).

Severity and intensity describe the physical change to ecosystem structure and the energy released during the process of disturbance, respectively (Oliver and Larson, 1996). These measures have little quantitative value for energy development because most activities are high-magnitude disturbances that tend to remove most aboveground biomass (Johnson and Miyanishi, 2008). The physical changes to ecosystem structure often do not include measures of the loss of ecological memory like stored seeds and buds in the soil profile, but these elements have important ramifications for the future structure of the ecosystem, water penetration, the ability of the soil to store carbon, and site productivity (MacFarlane, 2003).

Persistence describes the length of time the vegetation condition remains disturbed and is an important indicator of permanence. Anthropogenic disturbances are often characterized by high persistence, especially those related to energy resource development and linear corridors. Oil and gas activity often persists in the form of active well sites, pumping stations, and pipelines (Figure 2.1). Most linear corridors that are cleared in forests become *de facto* permanent access networks due to continued disturbance and repeated use by human activities (Lee and Boutin, 2006). As understood in the context of anthropogenic disturbance, persistence represents human control over successional accumulation of biomass and is a significant component of anthropogenic disturbance regimes (Odum, 1969). Forest harvesting occurs in such a manner as to promote the regrowth of trees at regular intervals, while linear corridors are maintained as right-of-ways for access purposes and energy development indefinitely constrains successional pathways (Lee and Boutin, 2006). Quantifying persistence in anthropogenic disturbance regimes will be a requisite for successfully understanding ecosystem dynamics of forests.

Figure 2.1 Aerial view of a typical hydraulic fracturing well site in the Marcellus shale basin, U.S. (Photo credit: Bill Howard—The Downstream Project, via LightHawk/SkyTruth).

Frequency quantifies the interval between successive disturbances at the same location. For most historical disturbance regimes, frequency is often inversely related to magnitude (*i.e.*, severity and intensity) (Johnson and Van Wagner, 1985), but anthropogenic disturbances related to energy development tend to be both frequent and high magnitude (Johnson and Miyanishi, 2008). Many land uses in forests are now related to energy extraction and these areas tend to undergo near-permanent land conversion to a non-forested state for an indefinite period of time (Lee and Boutin, 2006). Frequency is especially challenging to quantify for converted land because the land is not likely to return to a forested state during any ecologically-meaningful interval.

Seasonality describes the timing of a disturbance and this measure is important for characterizing anthropogenic disturbance regimes due to the variability in seasonal activities. For example, forest harvests are routinely undertaken during autumn or winter in consideration of breeding birds, soil damage, and erosion. Anthropogenic disturbances occurring during autumn or winter might favor invasive, opportunistic floristic species that disperse into a disturbed patch. Conversely, anthropogenic disturbances that occur in spring or summer might favor dominant floristic species because many tree species produce mast during this time in their lifecycle. Depending upon the timing and severity of disturbance, local ecological memory can be erased entirely and provide the means for invasive species to colonize the disturbed patch.

2.3 Ecological impacts of energy development

Energy development modifies disturbance regimes and landscape structure by clearing or modifying forest for non-renewable resources and thus can suppress historical disturbance regimes, introduce novel ecosystems, and eradicate ecological memory and biological legacies. Monitoring forest change due to oil and gas extraction requires an understanding of the impacts of such activities on the delivery of ecosystem services, which are summarized in Table 2.1, and discussed in the following subsections. I discuss the impacts of energy development that I regard as the most significant for forested ecosystems and the effects of which often elude mapping efforts predicated on discreteness.

Table 2.1 Examples of oil and gas impacts on ecosystem services.

Type of disturbance	Ecosystem services affected	Example	Reference
Surface mining	Carbon storage	Permanent peatland loss from oil sands mining in Alberta will release between 11.4 and 47.3 Mt C. Poor productivity for abandoned mining sites leading to a conversion from a carbon sink (forest) to a source (mine).	Rathore and Wright, (1993), Rooney *et al.* (2012)
	Water purification	Acid mine drainage and runoff, heavy metal contamination, and high concentrations of dissolved solids associated with coal mining. Seepage of heavy metals from more than 12,000 ha of tailing ponds associated with the oil sands mines into the Athabasca watershed, estimated at 11 million $L \cdot d^{-1}$.	MacFarlane (1999), Osborn *et al.* (2011), Rooney *et al.* (2012)
Pipelines	Erosion control Storm surge and flood buffering	Dredging of the Mississippi River delta plain for navigation and pipelines has resulted in significant loss of native marsh vegetation due to high influx of salt water, which has also increased erosion of the delta.	Dyer *et al.* (2001)
Seismic lines	Carbon storage	After 35 years, most seismic lines in Alberta had not recovered to a forested condition, thus reducing carbon storage of the forest.	Latham *et al.* (2011)
	Subsistence resources	Caribou (*Ranger* species) avoid linear features, presumably due to increased risk of predation by wolves, although the association between caribou mortality and distance to linear features like seismic lines remains disputed.	James and Stuart-Smith (2000), Sawyer *et al.* (2006), U.S. Department of Interior (2013)
Well sites	Water purification	Methane contamination of drinking water from hydraulic fracturing in the Marcellus and Utica shale formations of Pennsylvania.	Smyth and Dearden (1998)
	Carbon storage	Poor productivity of abandoned well sites and the conversion of a carbon sink (forest) to a source (producing well site).	Alberta ESRD (n.d.)
	Subsistence resources	Ungulates have been observed to avoid industrial features like well sites, thus resulting in functional loss of otherwise suitable habitat.	Timoney and Lee (2001), U.S. Department of Interior (2013)

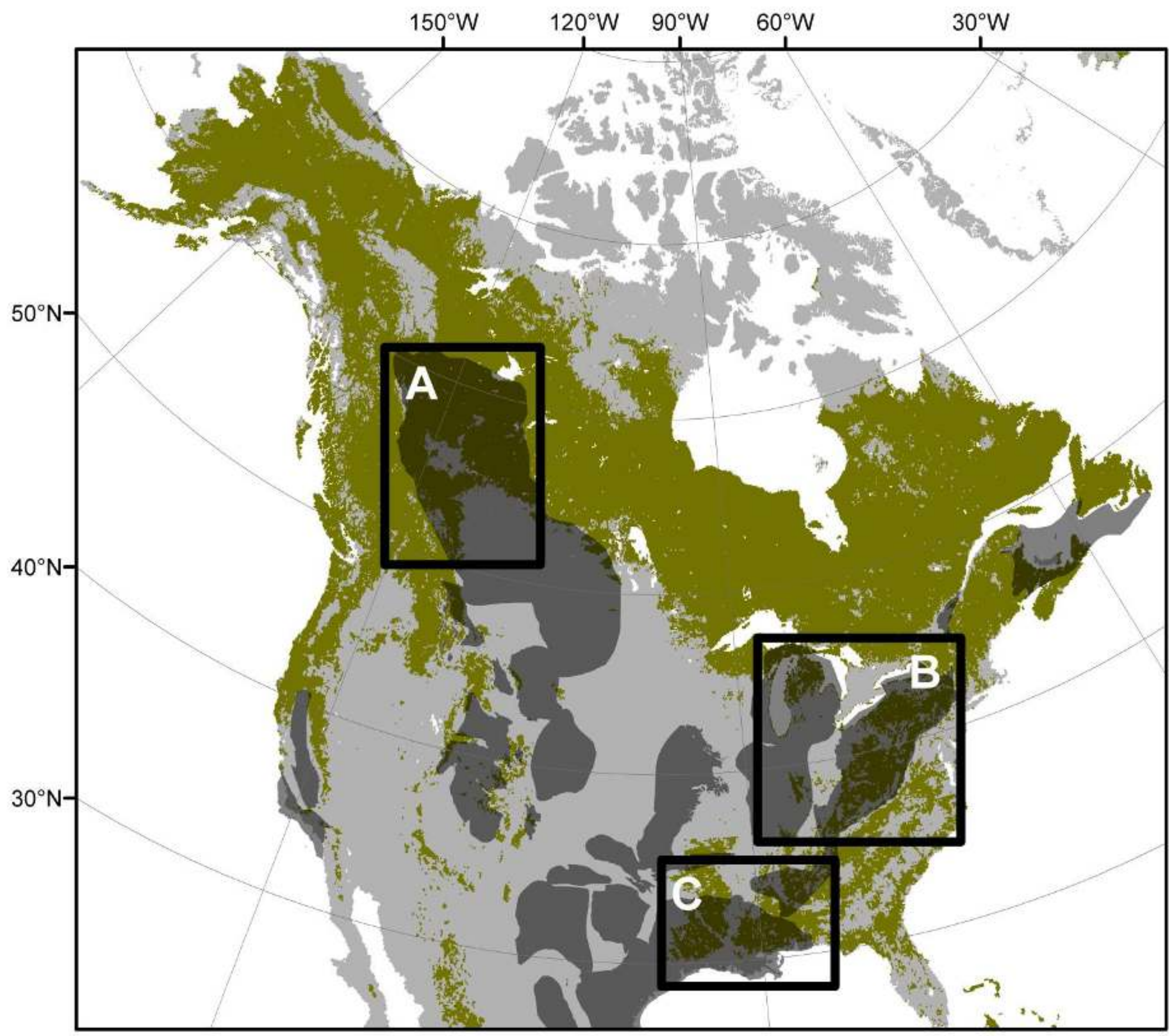
2.3.1 Clearing of forest for non-renewable resources

Non-renewable resources underlay many forests of North America (Figure 2.2) (Butt *et al.*, 2013). Historically, traditional surface mining was used to recover most non-renewable resources, which has conventionally been a high magnitude disturbance that can remove whole forest ecosystems and remains a primary land use for many forested regions in North America today. For example, coal is mined throughout Appalachia in the U.S.; uranium in Northern Saskatchewan, Canada; bitumen in Alberta's oil sands; diamonds and gold in Northwest Territories, Canada; and a variety of base metals in Ontario and Québec, Canada (Bernstein *et al.*, 2006). Nearly 140 million ha of boreal forest has been developed for oil sands mining in Alberta, more than one-fifth of the province (Johnson and Miyanishi, 2008). Approximately 338,000 ha of the Canadian boreal forest zone is directly disturbed by mines, oil and gas infrastructure, and well sites, exclusive of nearly 353,000 km of seismic exploration lines and pipelines that account for more than half of all linear features in the Canadian boreal (Pasher *et al.*, 2013).

In recent decades, several unconventional *in situ* technologies have been developed for recovering hydrocarbons and are being implemented around the continent. In the U.S., 90% of wells drilled on public lands, much of which is forested, use hydraulic fracturing (Figure 2.1) (U.S. Department of Interior, 2013). In Canada, a process known as steam-assisted gravity drainage is used to recover heavy and high-viscosity bitumen from deep oil sands deposits in Alberta. Many of these advancements in hydrocarbon recovery allow for a much larger sub-surface area to be developed, since deeper deposits can be recovered without surface mining. As a result, well sites related to hydrocarbon extraction have increased substantially on forested landscapes globally over the past two decades due to the rise in energy prices and advancements in recovery technologies.

Although well sites and mines are regularly mapped, many of their impacts are not discrete. Ecosystem services such as water purification, carbon storage, and subsistence resources can be negatively impacted by oil and gas activities (Table 2.1). For example, elevated levels of toxic elements by surface mining of the oil sands (Figure 2.2a), coal mining seep into local watersheds (Kelly *et al.*, 2010; Rathore and Wright, 1993; Timoney and Lee, 2009), and hydraulic fracturing in the Marcellus and Antrim shale basins (Figure 2.2b) has been

Figure 2.2 The coincidence of North American forest cover (green) and major underlying hydrocarbon basins (transparent gray). Black boxes bound forested regions currently undergoing rapid energy development: (A) oil sands in Alberta, Canada; (B) Marcellus and Antrim shale basins in northeastern U.S.; and (C) the Gulf of Mexico oil and gas basin in southeastern U.S.



associated with methane contamination of drinking water (Osborn *et al.*, 2011). Surface mining the oil sands and coal mining has been associated with massive carbon release (Rathore and Wright, 1993; Rooney *et al.*, 2012) and well sites and seismic lines can reduce carbon storage (Lee and Boutin, 2006; MacFarlane, 1999). Finally, forest habitat use by ungulates adjacent to industrial sites can be severely attenuated depending on the level of anthropogenic activity and changes to the distribution of these game fauna can severely impact both subsistence resources and ecosystem dynamics (Dyer *et al.*, 2001; James and Stuart-Smith, 2000; Latham *et al.*, 2011; Sawyer *et al.*, 2006).

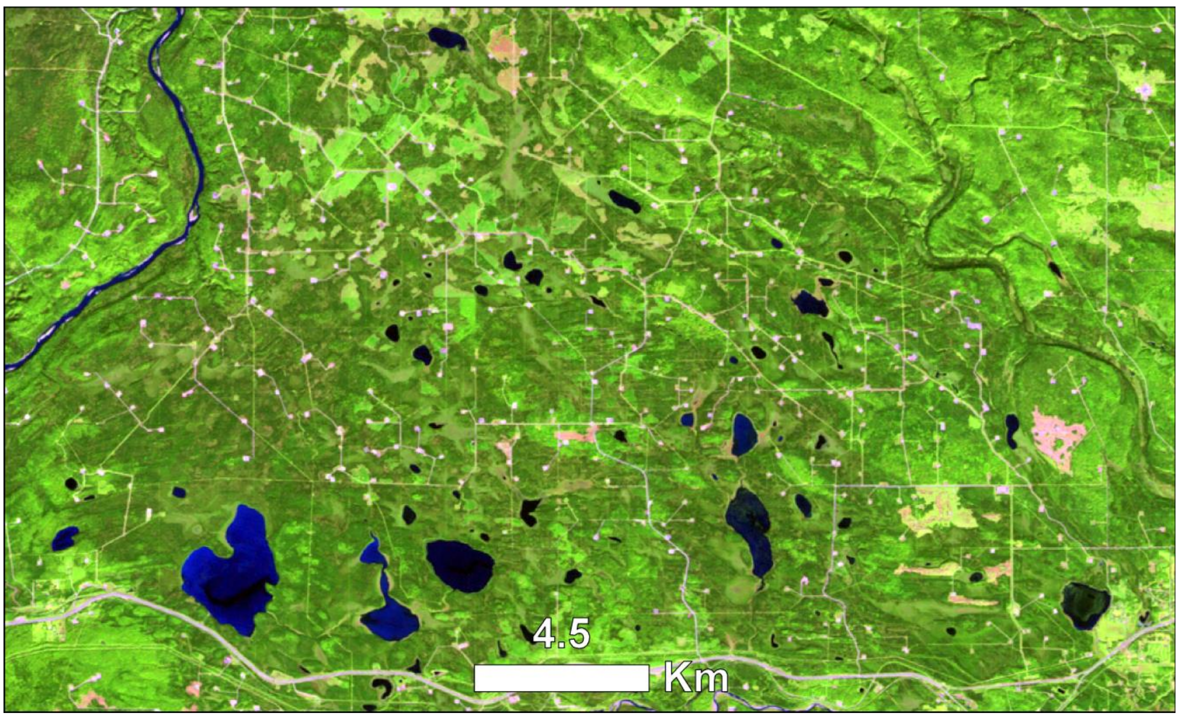
The legacies associated with non-renewable resource extraction can be significant barriers to ecosystem recovery. In North America, best practices and reclamation standards for non-renewable resource extraction are just as variable as monitoring and enforcement (Smyth and Dearden, 1998). The small size of individual well sites gives the impression that the footprint of the oil and gas industry is relatively small (Figure 2.3), however, approximately 400,000 well sites have been established in Alberta alone (Alberta ESRD, n.d.). Thus, the possibility remains that *in situ* extraction technologies degrade larger areas of forest than traditional surface mining. Detecting and quantifying the secondary impacts from non-renewable resource development will be critical areas of future research as these unconventional techniques become more commonplace throughout North American forests.

2.3.2 Persistent linear corridors

The clearing of forest for linear corridors represents one of the major challenges for conservation science in ecology. Linear corridors have at least three major impacts on forest ecosystems: access by people; travel by predators; and edge effects. Linear corridors directly remove trees and often understory vegetation to make travel easier through forested ecosystems. However, the impacts to vegetation and ecosystem dynamics often exceed the spatial boundary of the clearings (Dyer *et al.*, 2001).

Access into forests is one of the greatest impacts of forest land use. Most corridors are roads, but increasingly, many corridors are now related to the exploration and development of oil and gas (Lee and Boutin, 2006; Pasher *et al.*, 2013). In addition to the direct mining or extraction

Figure 2.3 False-color (bands 5-4-3) Landsat 5 image showing oil and gas well sites in Alberta, Canada.



site, ancillary infrastructures often accompany non-renewable resource development that might otherwise not be present on the landscape. For example, airports may be constructed for distant locations and railways, roads and pipelines for transporting the resources to market. Ecologically, the conversion of forest to access corridors could be considered a permanent change since any known recovery interval can exceed the lifecycles of many forest-dwelling species (Timoney and Lee, 2001).

Corridors related specifically to oil and gas infrastructure have increased significantly on the continent. An estimated US$29.1 billion will be spent annually over the next two decades on oil and gas infrastructure in both the U.S. and Canada (INGAA, 2014). Future pipelines are proposed to connect Northern Canada and U.S. refineries and ports in the U.S. Midwest and Gulf Coast (Parfomak *et al.*, 2013). Reflection seismology is also increasingly being used on forested landscapes more than during any prior time. Cut lines are cleared forest corridors used to explore subterraneous deposits of minerals and hydrocarbons beneath forests. Densities of cut lines on forested landscapes have reached as high as 2.7 $km \cdot km^{-2}$ in Alberta (Timoney and Lee, 2001). Like well sites, many cut lines can persist on landscapes without tree cover for more than three decades and many are converted to roads and other permanent access corridors (Lee and Boutin, 2006).

Linear corridors can modify ecosystem structure through predator-prey pathways. Travel and mobility are markedly enhanced by linear corridors resulting in increased predation rates (Sawyer *et al.*, 2006) and concomitant top-down trophic cascades on community structure (Hebblewhite *et al.*, 2005). Additionally, habitat use can be severely attenuated due to increased predation (Hebblewhite *et al.*, 2005; Leblond *et al.*, 2013; Lesmerises *et al.*, 2013, 2012). Many of these effects are not discrete and it can be challenging to characterize these anthropogenic impacts using traditional landscape pattern indices and patch-based statistics (Haines-Young and Chopping, 1996). For example, cut lines can have the largest footprint of any anthropogenic disturbance when a spatial buffer is applied (MacFarlane, 2003).

2.3.3 Suppression of historical disturbance regimes

Historical variability is the variation in ecological conditions at a specific scale (Landres *et al.*, 1999; Morgan *et al.*, 1994). Modern forest land uses suppress historical variability by abating the effects of historical disturbance regimes and homogenizing forest age classes. Throughout North America, intensive forest management zones are increasingly associated with non-renewable energy infrastructure (*e.g.*, pipelines, mines, well sites, pumping stations). The increasing presence of energy sector infrastructure has a marked impact on forest management such as continued fire suppression and mitigation efforts. The policy toward fire on forested lands throughout North America has been mostly reactive management (*e.g.*, extinguishing fires) rather than proactive management (*e.g.*, prescribed fire) (BC Wildfire Management Branch, 2010). A consequent legacy of this land use policy is the expenditure of approximately US$3.6 billion annually over the past decade, on average, for fire suppression efforts in North America and these costs are expected to increase in the future (Gorte, 2013; Gould *et al.*, 2013; Taylor *et al.*, 2006). In the U.S., the Smokey Bear campaign has enshrined reactive fire management as policy for over 60 years despite a general consensus among land managers that fire suppression has increased the costs and difficulty of suppressing fires (Donovan and Brown, 2007).

Throughout the continent, the success of fire suppression efforts has been mixed. In Alberta, an initial attack strategy decreased area burned by 457,500 ha between 1983 and 1998 (Cumming, 2005). In British Columbia, the largest exporter of forest products in Canada, initial attack ensures that 92% of fires are extinguished within 24 hours of detection (BC Wildfire Management Branch, 2012) and in the U.S. the success rate is between 97% and 99% (Stephens and Ruth, 2005). However, the success of suppressing the majority of fires is relative to fire size. The largest fires are the rarest, but comprise the majority of annual area burned, and area burned has actually increased over recent decades in U.S. despite rising suppression costs (Stephens and Ruth, 2005). Bridge *et al.* (2005) attribute discrepancies of successful fire suppression between regions to a lack of empirical evidence.

Characterizing the historical disturbance regime represents another area of controversy. Within the boreal zone, the characteristics of fire regimes vary spatially and temporally (Andison and McCleary, 2014). Without a complete understanding of the variability of the historical disturbance regime, it can be challenging to estimate exactly what is suppressed and to what degree. The lack of historical records limits any efforts of achieving such characterizations and most research has used empirical models of present-day fire behavior to estimate historical conditions (Andison, 2012).

Beyond the northern forests, offshore oil and gas activity in the Gulf of Mexico has contributed to wetland loss on the Mississippi River delta plain (Figure 2.2c). Dredging of the delta for shipping and navigation as well as undersea pipeline construction has resulted in significant loss of forested wetlands in the area due to the influx of salt water (Foy, 1990). These wetlands provide a number of regulating ecosystem services, including erosion control and storm surge buffering (Table 2.1, Barbier 2014) that are critical to the nearby city of New Orleans. Louisiana contributes approximately 12% of natural gas production in the U.S. and more than two times the total offshore production in the Gulf of Mexico (USEIA, 2013). Since Hurricanes Katrina and Rita in 2005, and the British Petroleum Deepwater Horizon oil spill in 2010, restoration of the wetlands has been recognized as a key priority, but the region continues to face significant challenges for implementing the restoration plans, in part due to the pre-existing oil and gas infrastructure (Barbier, 2011).

2.3.4 Introduction of novel ecosystems

Forest land uses can have dramatic direct and indirect effects on community structure. Access corridors are common throughout North American forests and these features can serve as vectors for transporting invasive species throughout a landscape (Meunier and Lavoie, 2012). Invasion of non-native species can result in changes to ecosystem composition, structure and function. Novel combinations of flora and fauna make the future state of a forested landscape challenging to predict and render comparisons to historical states unreliable. Non-native species can be introduced into forests directly by intentional planting, as in the case of Alberta where, up to *circa* 1990, cut lines and well sites were re-planted with non-native plant species (MacFarlane, 2003; Osborn *et al.*, 2011), or indirectly by unintentional transportation on vehicles and people (Mack *et al.*, 2000). Such dispersal mechanisms are exacerbated in forests undergoing significant energy development because density of access features can be as much as four times greater on these landscapes due to pipelines, cut lines, and roads connecting to well sites (Schneider *et al.*, 2003). Moreover, forest edge habitats are particularly prone to be invaded by non-native plant species due to exposure to wind, light, and nutrient availability (MacFarlane, 2003). The ecological impacts of non-native species are numerous and Ricciardi *et al.* (2013) review and compile hypotheses of the mechanisms related to each.

In North America, the distribution and abundance of predators has been altered by forest land uses related to energy development (Laliberte and Ripple, 2004). In the Rocky Mountain Foothills of Alberta, grizzly bear (*Ursus arctos*) populations have declined due to human-related mortality associated with the increasing density of access corridors in the region and the range of the species has been constrained by a greater presence of humans (Laberee *et al.*, 2014; Laliberte and Ripple, 2004). Wolves (*Canis lupus*) showed similar responses to human presence on heavily developed landscapes and, as a result, the predation success and behavior of the species was altered leading to other changes in the trophic web (Hebblewhite *et al.*, 2005).

Since the 1960s, predators have been perceived to exert top-down controls on community structure (Hairston *et al.*, 1960). Anthropogenic activity is believed to mediate trophic cascading by removing predators or increasing the rate of predation (Hebblewhite *et al.*, 2005). For example, in Yellowstone the expatriation of wolf was related to reduced cottonwood (*Populus* species) and aspen (*Populus tremuloides*) recruitment (Beschta, 2005; Ripple and Larsen, 2000). In Alberta, access corridors are thought to increase wolf predation on woodland caribou (*Rangifer tarandus caribou*) (Foy, 1990) and a similar observation was described for wolf distributions in Eastern Canada (Lesmerises *et al.*, 2013, 2012). However, a more recent study cast doubt on this mechanism (Kauffman *et al.*, 2010) and research involving other large predators has shown that top-down controls are not a panacea for ecosystem management (Marris, 2014). Nonetheless, such changes in predator-prey relations can have negative consequences for provisioning ecosystem services and subsistence resources like the distribution of game fauna (Table 2.1; Dyer *et al.*, 2001; James and Stuart-Smith, 2000; Latham *et al.*, 2011; Sawyer *et al.*, 2006).

Terrestrial arthropods are much less charismatic fauna than vertebrates, but represent the largest component of biodiversity in North American forested ecosystems and are responsible for the majority of predator-prey interactions. Due to their size and limited mobility, arthropods are particularly sensitive to changes in microclimate at local scales, which suits their application as ecological indicators (Langor and Spence, 2006). Species richness has been shown to increase in edge habitats throughout North America (Buddle *et al.*, 2006; Spence *et al.*, 1996; Van Wilgenburg *et al.*, 2001) and other boreal forests (Niemelä, 1997). Arthropod response to oil and gas activities is relatively understudied compared with forest harvesting (Buddle *et al.*, 2006; Spence *et al.*, 1996). However, research following the British Petroleum Deepwater Horizon oil spill in the Gulf of Mexico, during 2010, showed that terrestrial arthropods were quite sensitive to oil exposure, but retained the capacity to recover in the following season given adequate vegetation cover (McCall and Pennings, 2012). Arthropod response to natural gas flaring and terrestrial oil spills remains a critical area of future research given the rapid development of deep shale natural gas resources and offshore recovery technologies throughout the continent. Changes in community composition and trophic structure are very challenging to map because they are dynamic processes that are dependent on season-to-season conditions, yet these anthropogenic-mediated changes can have profound impacts on forested ecosystems.

2.3.5 Eradication of ecological memory

Many anthropogenic disturbances are high-magnitude events that remove the majority of woody biomass at the surface. Biological legacies such as seed stored in the soil profile and dead wood are termed ecological memory (Bengtsson *et al.*, 2003), so-called because they are the elements that allow an ecosystem to maintain its characteristic identity when disturbed. Forest land uses can severely damage the integrity of ecological memory by compacting or removing forest soils, removing the majority of woody biomass, and permanent or near-permanent conversion of forest to a non-productive state.

Surfaces such as roads and cut lines can severely modify or remove the underlying soil profile that stores the seed and bud potential of the forest (MacFarlane, 2003). If disturbance severely impacts the soil profile then colonization is likely to occur from dispersal rather than emerging from the local bud or seed bank. After 35 years, the majority of cut lines in Alberta continued to persist with a cover of low forbs and only 8% had recovered to a majority cover of woody vegetation (Lee and Boutin, 2006). Similarly, well site construction can severely compact soils and reclaiming these features to forest is limited by poor recruitment, growth, and survival by trees (USEIA, 2013). An apparent barrier to successful reforestation for these disturbances in Canada is the lack of clearly written standards and procedures for forests (MacFarlane, 2003).

Fires are known to leave patches of live island remnants and unburned residual forest within the perimeter of a burn (Andison, 2012). In the absence of stand-replacing fire, senescence of the pioneer forest cohort leads to large abundances of downed woody debris, which provides habitat for ground- and stream-dwelling fauna and colonization sites for emergent cohorts of autotrophs (Harmon *et al.*, 1986). Similarly, windthrow can catalyze localized patterns of forest regeneration (Mitchell, 2013). Total dead wood can comprise up to 18% of all woody biomass in cool, moist temperate forests (Krankina and Harmon, 1995, 1994). The practice of clear cutting significantly reduces dead wood compared to fire (Pedlar *et al.*, 2002) and coarse woody debris has been removed from stream and river channels in North America for centuries to improve navigation (Harmon *et al.*, 1986). The result of these activities has been complete anthropogenic engineering of forest and stream ecosystems and a significant loss of ecological memory related to dead wood structures.

The majority of forest land uses related to energy development permanently convert forest or remove forest cover for long periods, which can reduce the accumulation of ecological memory to forested ecosystems. For example, surface mining and activities related to non-renewable resource extraction can destroy decades-long accumulations of ecological memory stored in soils. A reclaimed coal mine grassland in Appalachia U.S. had significantly reduced levels of P compared with a reference watershed, and the macroinvertebrate community structure and in-stream leaf decomposition rates were particularly sensitive to the changes of forest cover (Simmons *et al.*, 2008). Moreover, seeding reclaimed forest land with native species has only become accepted in the U.S. (Richards *et al.*, 1998) and Canada (Timoney and Lee, 2001) since the 1990s. Ecological memory can be very difficult to map aerially since the majority of these fine scale structures occur below the ground surface. A great challenge remains for mapping and studying the impacts of forest land uses on the distribution, abundance, and quality of ecological memory.

2.4 Detecting anthropogenic disturbance regimes

Central to remote detection of disturbance is the concept that energy absorption and reflectance varies with the structural characteristics of land cover types. Remote detection of vegetation disturbance such as forest relies on energy exchange properties in several key regions of the electromagnetic spectrum: (1) the absorption of visible energy; (2) the scattering of red and near-infrared energy at the "red edge"; (3) the absorption of short-wave infrared energy; and (4) the reflection of thermal energy (Nemani and Running, 1997; Rock *et al.*, 1986). Property one is associated with leaf pigments, properties two and three are associated with cell structure, and property four is associated with the fractions of soil and vegetation in a pixel (Nemani and Running, 1997). Nemani and Running (1997) were among the first to recognize a disturbance trajectory by relating energy exchange to land cover types.

The initial work on relating properties of energy exchange to land cover types pioneered a number of breakthroughs in disturbance detection specific to forested ecosystems. Healey *et al.* (2005) developed a disturbance index (DI) based on a linear combination of Kauth and Thomas' (1976) earlier tasseled cap transformation of Landsat-derived spectral reflectance. With the free public release of Landsat imagery in 2008 (Woodcock *et al.*, 2008), and significant advancements in computing capabilities, every satellite observation in the Landsat archive is now being utilized for change detection and classification of land cover (Wulder and Nelson, 2003; Zhu and Woodcock, 2014). Automated change detection has now been made possible by novel algorithms such as the object-based disturbance inventory framework (Linke and McDermid, 2012, 2011; Linke *et al.*, 2009), the vegetation change tracker (VCT) (Huang *et al.*, 2010), and Landsat-based detection of trends in disturbance and recovery (LandTrendr) (Kennedy *et al.*, 2010), which have been used to detect forest change at continental scales for the first time using Earth observations (Masek *et al.*, 2013). Such advancements in the discipline and computer processing have led to the first high-resolution map of recent forest cover changes for the entire planet (Hansen *et al.*, 2013) and near-real time monitoring of forest condition (U.S. Forest Service, n.d.).

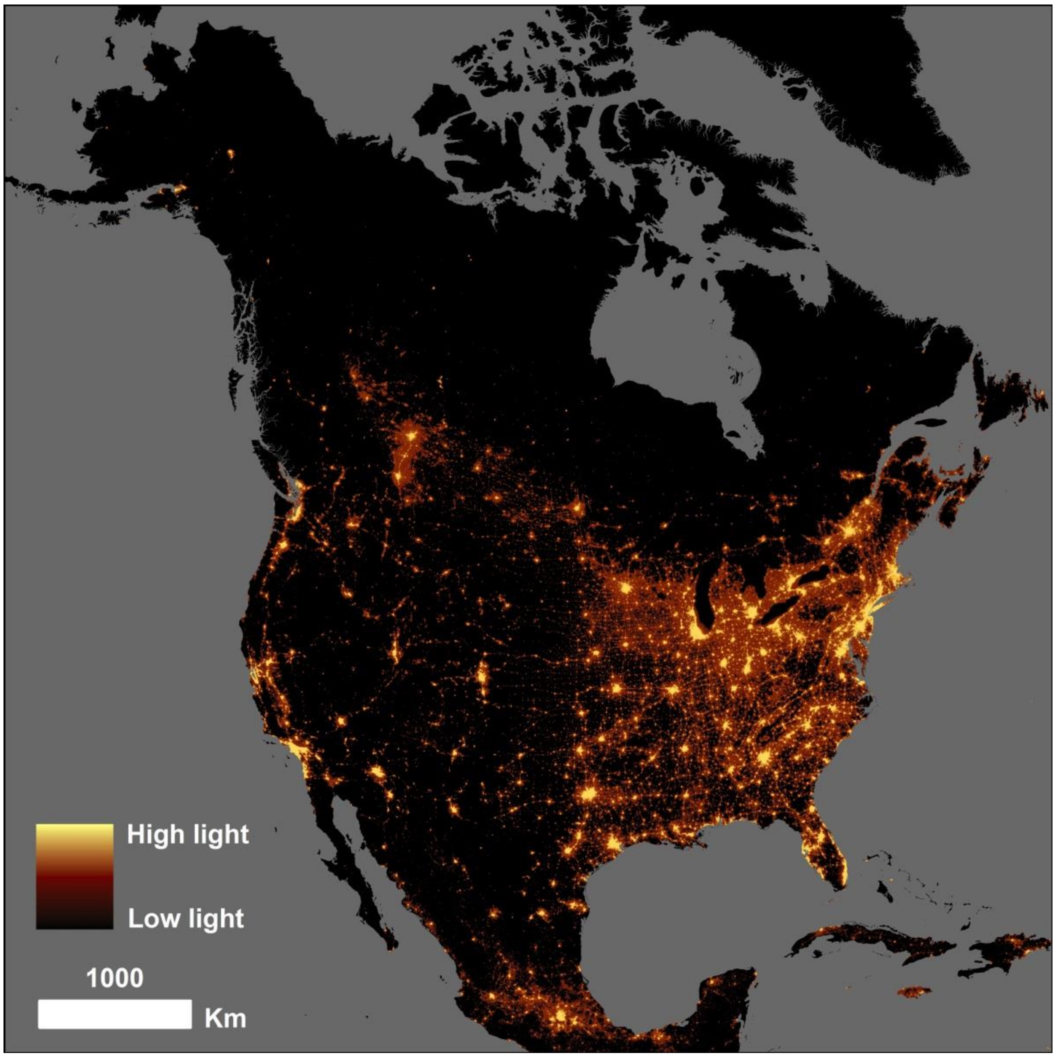
To date, there have been a number of studies focused on detecting, mapping, and monitoring forest change related to energy development. Fernández-Manso *et al.* (2012) showed that coal mines could be extracted and mapped as features derived from endmember spectral mixture analysis using Landsat data with consistent and accurate results across three global study areas. Townsend and colleagues (2009) characterized changes in the extent of surface mining and reclamation in Appalachia U.S. and observed that forest cover and active mines declined between 1976 and 2006 while reclaimed land increased over the same period. Such detection and mapping approaches can complement the disturbance inventory framework proposed by Linke *et al.* (2009), and are well-suited to multiple-use landscapes undergoing rapid energy development.

Change detection at finer scales will be essential for contextualizing and attributing disturbance impacts to forest land uses. While there has been tremendous advancement in remote detection of disturbance and forest change using known spectral properties of vegetation, there has been less progress toward developing new methods for detecting novel types of change seen in the Anthropocene. Ellis and Ramankutty (2008) methodically mapped the relationship between vegetation cover derived from Moderate Resolution Imaging Spectroradiometer (MODIS) and population density on a global scale to develop the anthromes concept. In the future, it will become necessary for research programs to detect and monitor change at finer spatial scales since the impacts of anthropogenic disturbance to forests are most commonly more localized and heterogeneous than a Landsat or MODIS pixel represents. Aerial photography is the oldest remote sensing technology that has resulted in the longest continuous record of Earth observation (Morgan *et al.*, 2010). Recent advancements in automated classification of aerial photographs may open the possibility of interpreting the historical caches of aerial photographs and make change detection at finer scales practical for some regions (Morgan and Gergel, 2013).

Reliable detection of ecosystem structure around access corridors will permit sound characterization of ecological response to anthropogenic disturbance. Light detection and ranging (LiDAR) has emerged as a means to directly sense—rather than infer from optical wavelengths—forest structure. Discrete return LiDAR systems (small footprint LiDAR) operate at a spatial scale of approximately 0.2 to 0.9 m and full waveform systems (large footprint LiDAR) generally detect structures above 8 m (Lim *et al.*, 2003). Discrete return systems are useful for detecting structural information at the scale of an individual tree while full waveform systems can detect the variability of structure height at a plot scale. Airborne scanning LiDAR has been applied to measure a variety of structural attributes of forests including canopy height, mean stem diameter, stem number, and stem volume (Næsset, 2002). LiDAR may be particularly robust for detecting structural forest change, especially for regions that are disturbed with finer-scale access corridors such as roads, cut lines, pipelines, and transmission lines. Full waveform systems have the potential to detect and characterize anthropogenic edge features that often dominate anthropogenic landscapes.

Detection of anthropogenic disturbance regimes will require the use of multi-scalar Earth observations. Combinations of spectral, contextual and structural information will lead to new breakthroughs in processing, analyzing and characterizing remotely sensed data. There remain many possibilities for detecting anthropogenic disturbance regimes. For example, nighttime light emittance by human settlements and other activities is a fascinating and underexplored field of investigation (Small *et al.*, 2011), especially for detecting the spatial influence of anthropogenic disturbance regimes (Figure 2.4). Nighttime light emittance was recently used to estimate global natural gas flaring (Elvidge *et al.*, 2009), which could be particularly relevant to some North American forests where non-renewable resource development is expanding. Ultimately, Earth observation and characterization of anthropogenic disturbance regimes should aim to detect energy exchange via a range of spectral, spatial, and temporal domains. The importance of direct observation, such as *in situ* long-term monitoring should not be understated. Ground surveys and studies are essential to supplement, train, and verify remote sensing techniques. Developing research should focus on relating ground data to spectral signals that can be applied across broad scales for monitoring purposes. The future development of a framework for relating remotely-sensed indicators to ecological and social responses will be a capstone in the progress toward monitoring the Anthropocene.

Figure 2.4 Nighttime light intensity for North America collected in 1996 and 1997 by the Operational Linescan System of the Earth-observing Defense Meteorological Satellite Constellation.



Chapter 3

Characterizations of anthropogenic disturbance patterns in the mixedwood boreal forest of Alberta, Canada

3.1 Introduction

Many believe that ecosystem-based management (EBM) approaches to policy and land planning are important for advancing sustainable forestry practices (Seymour and Hunter, 1999). One of the methodologies hypothesized to achieve the intended outcomes of EBM in the boreal forest is to use disturbance patterns as guides for harvest planning (Hunter, 1993). This principle posits that anthropogenic disturbances that approximate the historical range-of-variability (HRV) of ecosystem patterns, processes and conditions are more likely to maintain higher levels of biodiversity (Buddle *et al.*, 2006; Levin, 2000), habitat connectivity (Saunders *et al.*, 1991) and habitat heterogeneity (Turner, 1989) across a landscape relative to traditional value-optimization land management. Range-of-variability approaches to forestland management have emerged as a means to characterize the variability of landscape pattern that results from disturbance (Bergeron *et al.*, 2007, 2001; Landres *et al.*, 1999). These analysis strategies provide statistical context for evaluating the current configuration of landscape pattern and, more recently, disturbance patterning (Andison, 2012). In order for HRV initiatives to be successful, land managers and forest planners need information and tools that can help them (1) understand the dominant disturbance processes and patterns that were historically responsible for maintaining ecosystem function and landscape pattern prior to industrialization, and (2) how the current disturbance process and patterns differ.

An HRV methodology is theoretically well-suited to the boreal forests of Canada. Across most of the Canadian boreal forest, stand-replacing fire is the dominant disturbance agent responsible for forest structure and patterns observed on these landscapes. Although the majority of fires are smaller than 1,000 ha, the largest 1% of lightning-caused fires account for 98% of the total area burned (Cumming, 2001a). Mortality levels within boreal fires are great enough to replace the previous forest with a new cohort (Johnson, 1992), although recent evidence suggests that boreal fires leave behind a large range of live vegetation (Andison, 2012). At finer spatial scales, there is evidence that forest stand composition influences burning patterns (Cumming, 2001b), and the median size of unburned live residuals increases with fire size (Eberhart, 1986).

Boreal fire characteristics have already influenced harvest regime patterns. Until recently, forest harvesting in Alberta was required to be practiced under a multiple-pass, clear cut dispersed design where vegetation is removed from similarly-sized harvest units (McRae *et al.*, 2001). This harvesting design results in forested landscapes often associated with "checkerboard" landscape patterns (Delong, 2002; Franklin and Forman, 1987). After a period of 10 to 15 years, the remaining forest adjacent to cut blocks is harvested. Dispersed harvest designs have been shown to increase edge density (Franklin and Forman, 1987), reduce interior old forest area, create significant road networks with associated increases in ecological and economic costs, reduce snag density and reduce forest patch size (Andison and Marshall, 1999; DeLong *et al.*, 2004), all of which result in negative biological consequences (Buddle *et al.*, 2006; Van Wilgenburg and Hobson, 2008).

In response to these ecological and economic concerns, forest management has been shifting away from dispersed harvest designs over the last decade in Alberta (Work *et al.*, 2003) and beyond (Andison and Marshall, 1999; Cyr *et al.*, 2009; Hunter, 1993; Nonaka and Spies, 2005) in favor of harvest designs premised on fire disturbance characteristics. These harvest designs aggregate harvest units into single-pass events of a range of sizes, composed of a mosaic of disturbed and undisturbed vegetation of various sizes, shapes and proportions (Carlson and Kurz, 2007; Dzus *et al.*, 2009) with the explicit aim of reducing the unintended negative outcomes of traditional harvesting pattern on boreal mixedwood landscapes (Van Wilgenburg and Hobson, 2008).

Consistent with EBM theories, the rationale for an aggregate harvest design is that organisms that have adapted to the spatiotemporal patterns generated by the historical fire regime are presumed to be more successful at persisting on the aggregate harvest landscape due to the higher variability and availability of habitat types (Merriam and Wegner, 1992). More specific to the boreal, the increased heterogeneity of landscape structure and pattern as a result of anthropogenic disturbances modelled after historical fire patterns is also presumed to maintain ecological adaptive capacity against unintended regime shifts, including climate change impacts

(Drever *et al.*, 2006). Furthermore, the empirical evidence in the boreal mixedwood supports this hypothesis (*e.g.*, Van Wilgenburg and Hobson, 2008). However, the efficacy of aggregate harvest designs largely remains an untested hypothesis. First and foremost, the degree to which aggregate harvest efforts approximate historical fire disturbance patterns is largely unknown. As Andison and Marshall (1999) found, the desire to emulate fire patterns does not necessarily translate so easily into practice. Perhaps of greater concern is that forest harvesting is only one of many stakeholders operating on boreal forested landscapes. Alberta is a provincial jurisdiction of Canada endowed with significant forest and energy resources (*e.g.*, oil, natural gas, and coal). The management of these natural resources over the last several decades has generated a complex spatial arrangement of anthropogenic disturbances across the forested landscapes of Alberta. The efforts of forest management agencies to emulate historical landscape patterns may be compromised by the cumulative effects of other stakeholders (Paine *et al.*, 1998; Schneider *et al.*, 2003; Timoney and Lee, 2001; Timoney, 2003).

To address these questions, disturbance patterns of a landscape in central Alberta that were attributed to forest harvesting and energy extraction were investigated and evaluated for divergences from a regional HRV based on known fire patterns. Based on the works of Burton *et al.* (2008), Carlson and Kurz (2007), Delong and Tanner (1996), and Delong (2002) it was hypothesized that current anthropogenic disturbance patterns such as total area in remnants, largest disturbed patch area (LDPA), number and size of island remnants are outside the HRV for the mixedwood boreal forest of northern Alberta. It was further hypothesized that the relative gap between current resource extraction and historical fire patterns varies significantly by the type of anthropogenic disturbance (*e.g.*, forestry vs. energy sector and dispersed vs. aggregate harvest). The hypotheses were tested on a large forest management area in the boreal mixedwood forest of Alberta.

3.2 Materials and methods

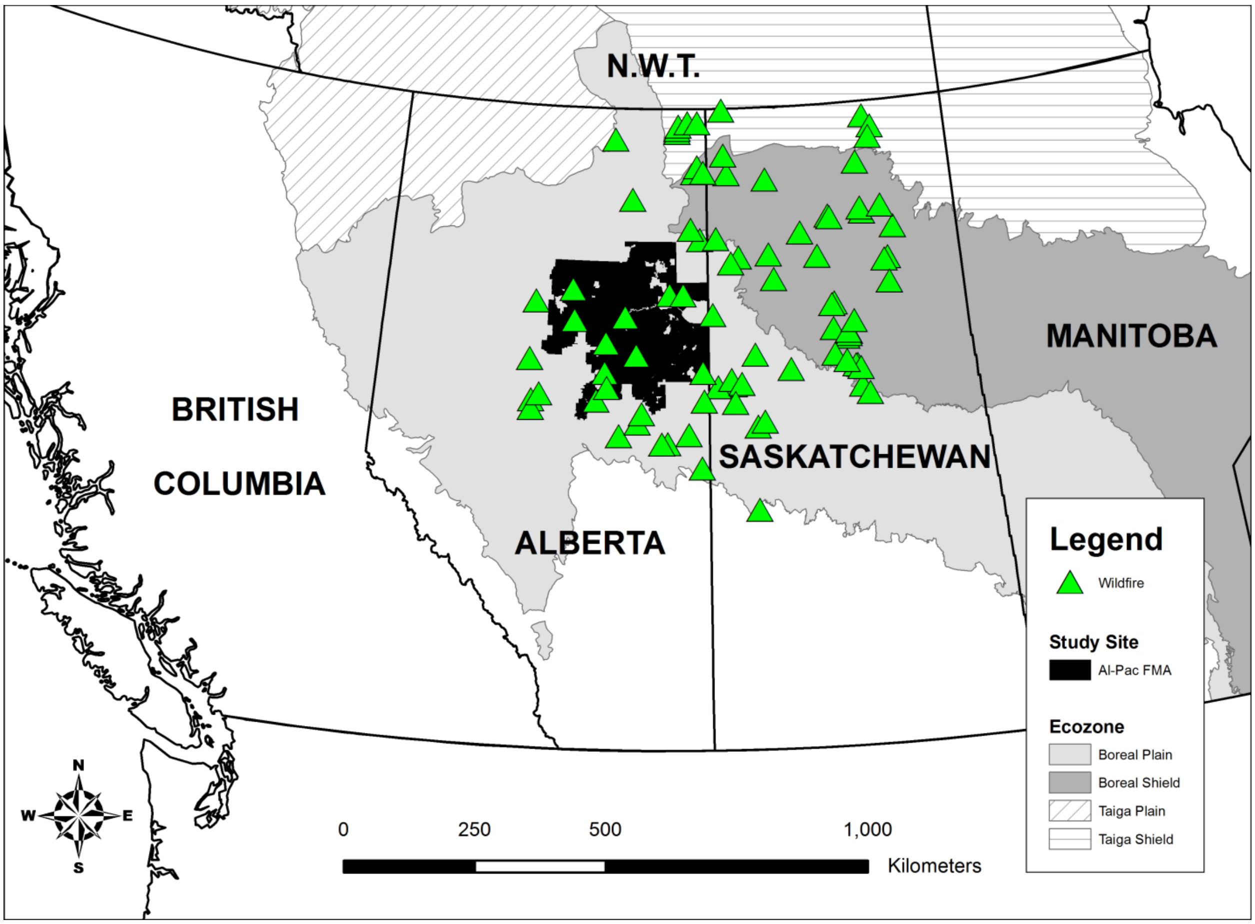
3.2.1 Study area

The Alberta-Pacific Forest Industries Inc. (Al-Pac) forest management agreement area (FMA) is 5.8 million ha in size and located wholly within the Boreal Plains, a bio-geographical ecozone

(ESWG, 1995) in north-central Alberta (56°21'N, 112°22'W) (Figure 3.1). The Boreal Plains ecozone is a national (Canadian) demarcation of a regional ecological system characterized by a continental interior climate, mostly flat undulating terrain and is composed of mixedwood, coniferous and deciduous forests (ESWG, 1995). Approximately half of the Al-Pac FMA is covered by wetlands of various types, many of which are forested (Alberta-Pacific Forest Industries, 1999). In central Alberta, the mean monthly temperature ranges from -16° C in January to 16° C in July (Strong, 1992). Total annual precipitation is approximately 400 mm of which approximately 60 mm is derived from snowfall (Strong, 1992). The topography exhibits limited variation in landforms with elevation ranging from approximately 300 to 900 m.a.s.l. Mesic sites tend to be dominated by *Populus tremuloides*, *Picea glauca*, and *Populus balsamifera* mixedwood stands; xeric sites tend to be dominated by *Pinus banksiana*; and forested wetlands are dominated by *Picea mariana* and *Larix laricina*.

Fire is by far the most common and prominent disturbance agent in the Al-Pac FMA, and thus serves as the foundation for the HRV estimates. Other endogenous disturbance agents include windthrow and forest tent caterpillar (*Malacosoma disstria*), but both create very fine scale disturbances on a far less frequent basis and are associated with relatively low levels of mortality (Roland, 1993). The current range of variability (CRV) represents the variability of anthropogenic disturbance patterns following human industrialization in the region since the 1940s. Since the late 1940s, sources of anthropogenic disturbance to the Al-Pac FMA include extensive energy exploration near Fort McMurray, the extraction of bitumen from the associated bituminous sands via open pit mining and *in situ* operations, natural gas extraction, timber harvesting, agricultural land conversion, town settlements, and linear right-of-way disturbances such as railways, roadways, seismic lines, transmission lines and pipelines. Prior to the inception of the FMA in 1993, forest management was primarily characterized by multi-pass, dispersed harvesting. Since *circa* 2000, Al-Pac has shifted their timber harvesting planning toward single-pass, aggregate harvest designs (Dzus *et al.*, 2009). Anthropogenic disturbances attributed to forest harvesting and energy extraction were both included in the CRV.

Figure 3.1 Map of reference fires occurring within ecozones of western Canada.



3.2.2 Historical fire data

Previous analysis of 87 aerially-interpreted fires with no record of suppression (Andison and McCleary 2014) provided a baseline for HRV disturbance patterns. All fires occurred between 1948 and 2004 with a geographic extent spanning from 110°0'W to 120°0'W and from 54°42'N to 60°1'N within the Boreal Shield, Boreal Plains, Taiga Plain and Taiga Shield ecozones of Canada, and were mapped to a minimum polygon size of 2 ha (Figure 3.1). As detailed in Andison and McCleary (2014) the criteria used to select unsuppressed fires for analysis had an upper size limit of 30,000 ha and therefore does not reflect the historical distribution of fire size.

3.2.3 Anthropogenic disturbance data

Sampling a FMA presents several challenges. First, the shape is irregular and, in the case of the Al-Pac FMA, there are several areas of unmanaged forest within the tenure. Additionally, different regions of the Al-Pac FMA are managed to different intensities based on regional stakeholders (*e.g.*, First Nations, energy companies) and timber quality. In terms of the spatial patterns of disturbance, it was assumed that harvest pattern and extent was similar over the entire FMA. Therefore, data to represent the CRV were obtained from a sub-sample of four areas within the Al-Pac FMA, each 12 townships (approximately 112,000 ha) in size (Figure 3.2). These four sample areas were chosen to represent the range of CRV sources and impacts across the FMA and were qualitatively selected for aggregate and dispersed harvest disturbances by staff at Al-Pac (see Table 3.1). Further analysis of site conditions for each sample area found that forest cover accounted for between 78% and 85% of each total land base. *Populus tremuloides* and *Picea mariana* are the dominant forest species across all sample areas, accounting for between 58% and 86% of each total forested land base.

For each of the four sample areas, spatial datasets representing the disturbance activities of the forestry and energy sectors were created as follows. First, the most recent (*circa* 1999) aerially-derived Alberta vegetation inventory (AVI) data, mapped at a minimum polygon size of 2 ha, were obtained (Alberta ESRD, 2005). All major highways, well sites, pipelines, permanent water bodies and forest harvest polygons were extracted from the AVI dataset. Second, the most

Figure 3.2 Map of sample areas within the Al-Pac study area.

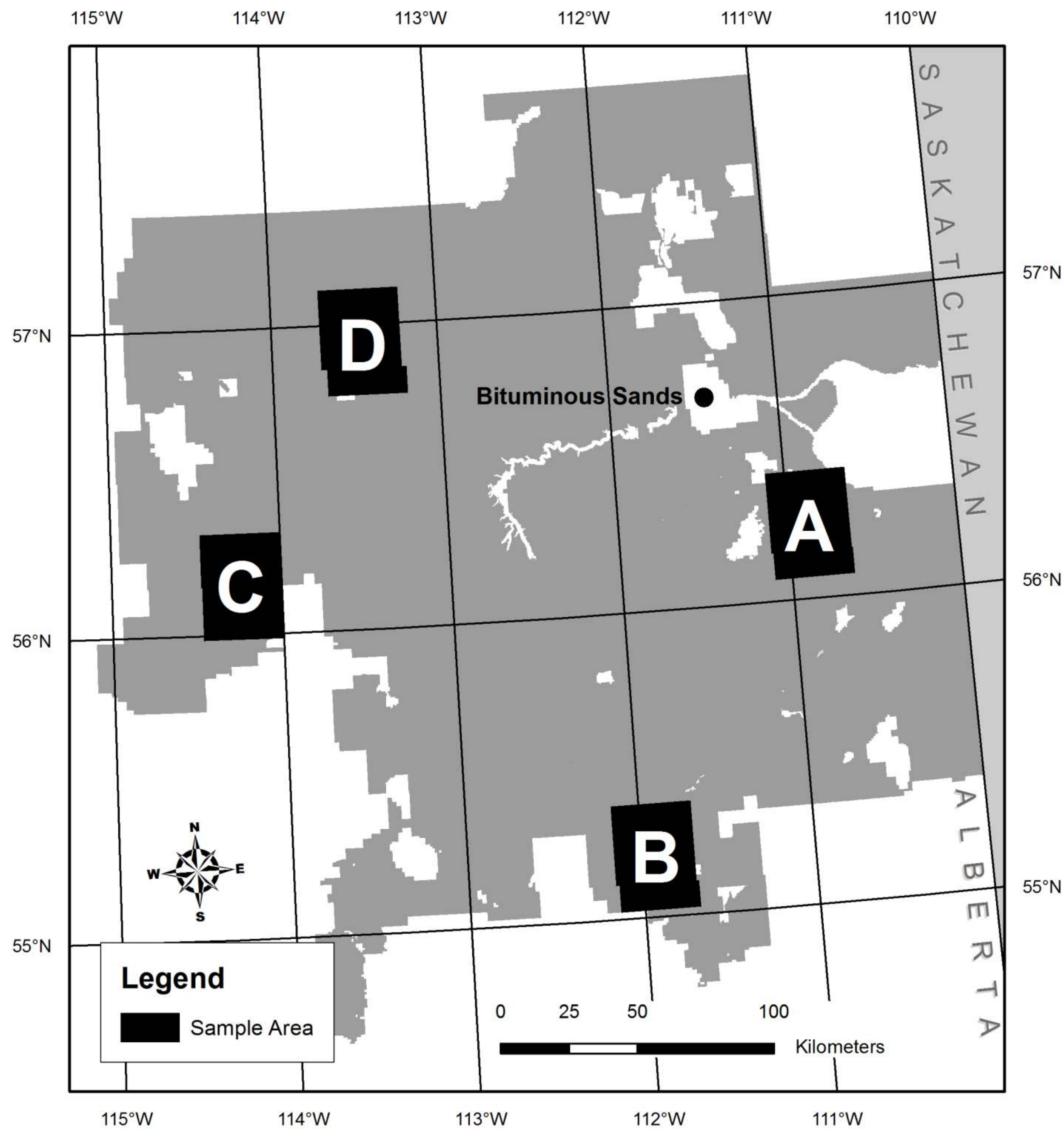


Table 3.1 Approximate areal contributions of different sources of disturbance to the land base of each sample area derived from the Alberta Vegetation Inventory (AVI) dataset.

	Study Area			
Disturbance Source	A (ha)	B	C	D
Forestry industry	9,140.7	8,276	2,913.6	855.2
Energy industry	1,343.4	602.9	182.9	426.7
Agriculture	0	9.8	0	0
Other industry	986.7	44.7	43.3	59.4
Fire	2,728.6	15,630.4	11,827.3	320.7
Windthrow	71.5	20.1	58.7	2.4

recent spatial data for seismic lines, utility lines, roads, railways and timber harvesting were sourced from Geographic Information System (GIS) databases at Al-Pac. To avoid bias associated with the sample areas, all disturbance events that partially fell outside of the sample areas were excluded from analysis. Linear features, such as roads and seismic lines, were clipped at the sample area borders.

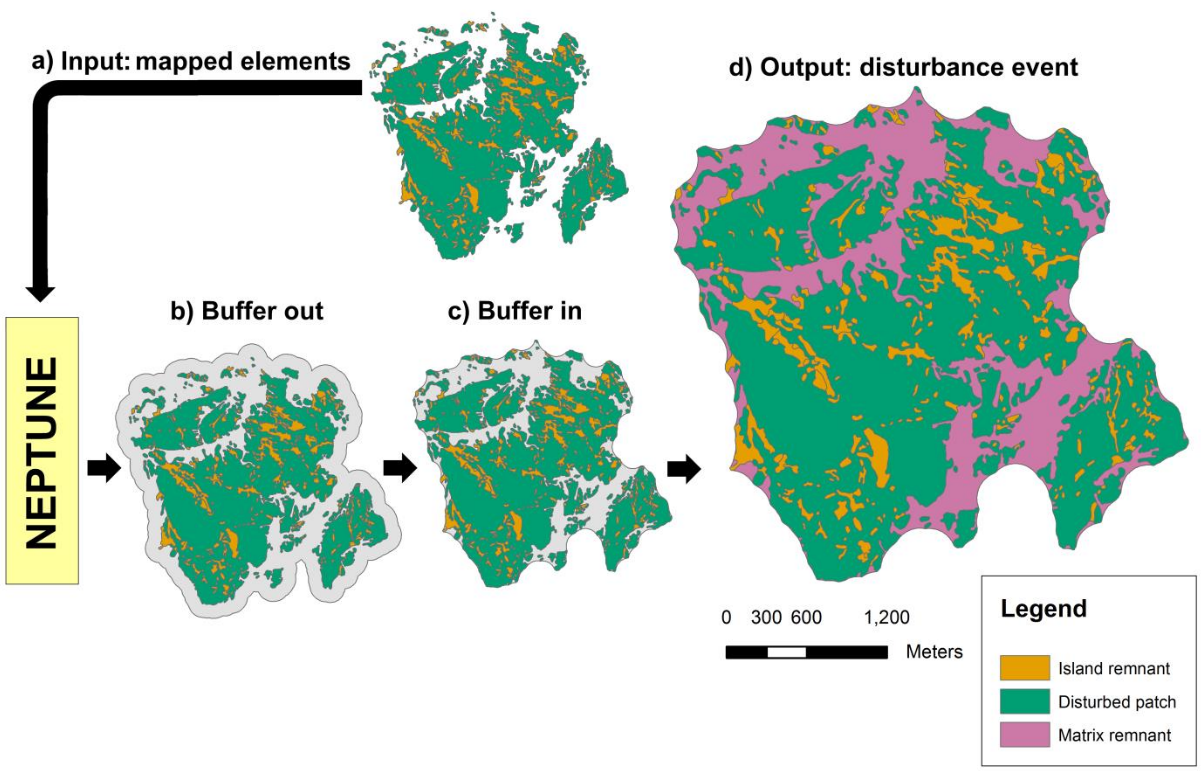
3.2.4 Disturbance pattern language

A challenge with characterizing disturbance patterns is being consistent and unambiguous with the selected pattern terms and spatial language (White and Pickett, 1985). Subtle differences in the delineation of fire boundaries can create significant differences in fire patterns (Andison, 2012), which can ultimately create regulatory and credibility challenges. The disturbance pattern language developed by Andison (2012), which is predicated on the concept of the general area of influence of a fire (*sensu* Burton *et al.*, 2008), was adopted for analyzing the spatial layers.

This spatial language involves both mapped and generated spatial elements, which together delineate spatially-discrete and objectively-defined disturbance events (Andison, 2012). The mapped elements derived from aerial interpretation include both disturbed patches (*i.e.*, >95% crown mortality) and island remnants. A buffering process takes the mapped spatial elements (*i.e.*, islands and disturbed patches; Figure 3.3a) and buffers them by user-defined spatial and temporal parameters (Figure 3.3b). The final step is to apply a negative buffer (or nibble) of the same distance to the mapped spatial elements (Figure 3.3c). These two steps of expansion and contraction generate a third type of spatial element known as matrix remnants, which occur among disturbed patches (Figure 3.3d).

This process aggregates disturbed patches (*e.g.*, harvest areas) that are close in space and time and delineates a single spatial entity known as the disturbance event, which consists of the input spatial layers, the generated matrix remnants and the generated disturbance event boundary. This three-level spatial language was adopted because it represents historical burning patterns and processes more precisely than the traditional binary system originally developed by (MacArthur and Wilson, 1967) which is predicated on the concept of habitable “islands” surrounded by an inhabitable “matrix”.

Figure 3.3 Diagram of the NEPTUNE (Novel Emulation Pattern Tool for Understanding Natural Events) process for delineating disturbance events.



3.2.5 NEPTUNE decision-support tool

Disturbance patterns were generated for historical fires and anthropogenic disturbance data using the web-based NEPTUNE (Novel Emulation Pattern Tool for Understanding Natural Events) decision-support tool (TFC, 2009). NEPTUNE combines the spatial language described in section 3.2.4 with the HRV of fire patterns as described in section 3.2.2. NEPTUNE uses a spatial and temporal buffer to merge disturbed patches into disturbance events (as described by Andison, 2012). All disturbance polygons were assigned the same year of disturbance (which allows NEPTUNE to consider all neighboring polygons). The spatial and temporal parameters of the spatial language have the flexibility to be scaled in NEPTUNE depending on the nature of the disturbances on the landscape. A 200 m buffering threshold was selected because Andison (2012) found that, at this distance for boreal landscapes, most disturbed patches of fires were correctly aggregated into their respective disturbance events and it was not too distant that patches from other disturbances were incorrectly aggregated. According to the spatial language, roads are treated as fully disturbed polygons, and, although they are not used in the creation and delineation of disturbed patches, roads are used to identify island remnants that are wholly surrounded by road or that lay between a disturbed patch and a road. As a result, roads, which occur within an anthropogenic disturbance event, may aid in the creation of island remnants among disturbed patches.

3.2.6 Disturbance event pattern indices

A suite of metrics that characterize disturbance events can be derived from NEPTUNE including event area, disturbed patch area and percentage, matrix remnants area and percentage, island remnants area and percentage, total remnants area and percentage, number of disturbed patches, number of island remnants, edge density, largest disturbed patch area and percentage, largest island remnant area, and mean island remnant area.

3.2.7 Data pre-processing

Seismic lines were buffered using ArcMap 10 (ESRI, 2010) by the width defined within the dataset, which ranged from 1 m to 8 m. When a seismic line feature lacked a width identifier, the arithmetic mean width value of all seismic lines (2 m) was used, which was applied to 147 features representing ~5% of total seismic line length in the study area. Utility lines and road features were buffered by 50 m to either side of the linear feature representing the average width (100 m) of right-of-ways for major features in Alberta. Unimproved roads and trails were excluded from analysis because the right-of-way width could not be accurately determined. Dispersed harvests (*e.g.*, harvesting that occurred before the year 2000) were analyzed separately from aggregated harvests (*e.g.*, harvesting that occurred after the year 2000). The AVI data and fire history records revealed no instances of fire in the selected harvested areas.

3.2.8 Statistical analysis

The non-parametric Kolmogorov-Smirnov (K-S) test was used to assay statistical differences between the cumulative distribution functions (CDF) of each disturbance source to the empirical cumulative distribution function (ECDF) provided by the HRV. The Mann-Whitney-Wilcoxon (MWW) rank sum test was used to assay significant differences between the location parameters (median) of anthropogenic treatments and HRV. The K-S test statistic D is a measure of the greatest observed vertical departure between any two CDF. Therefore, the K-S test is useful for analyzing differences between two distributions that do not meet the assumptions required for more common parametric analysis. The MWW test statistic U is the sum of all the ranked differences for each observation between any two treatments. The MWW test is useful for indicating differences in location between two similarly-shaped samples with positively skewed data, and, for large sample sizes, the distribution of U approaches a normal distribution. Both tests were performed in R (R Development Core Team, 2011) as two-tailed with significant relationships at $\alpha = 0.05$.

3.3 Results

3.3.1 Historical fire patterns

The median area of historical fires in island remnants was 22%, compared to just 6% for matrix remnants (Table 3.2). All of these fire disturbances were composed of fewer than 22 total disturbed patches and LDPA accounted for 88% of the disturbed area at the median and 85% on average. The mean size of disturbed patches was 247 ha at the median and 762 ha on average, which suggests the data are positively skewed. Mean island remnant area was 32 ha on average and 9 ha at the median. The largest single island remnant was 459 ha on average and 28 ha at the median. The number of island remnants associated with historical fires ranged from zero to 272, where the median and mean were equivalent to seven and 20, respectively.

3.3.2 Anthropogenic disturbance patterns

From all of the input spatial layers, 15,431 anthropogenic disturbed patches were identified. The NEPTUNE analyses of the spatial layers for the study area delineated 1,355 anthropogenic disturbance events. Of those, 126 were forestry-related disturbance events generated from 1,210 forest harvest polygons and 1,057 were energy-related disturbance events generated from 1,504 well site polygons. Forest harvest areas were further resampled by two harvest designs: 98 disturbance events were generated from 707 dispersed harvest pre-2000 polygons and 74 disturbance events were generated from 722 aggregate harvest post-2000 polygons. The resampling of forestry-related disturbed features by harvest designs generated more disturbance events than the forestry sector overall because the spatial arrangement of forestry features (*i.e.*, harvest areas) varied across the landscape by harvest design, resulting in different delineations of disturbance events using the same set of disturbed features.

3.3.3 Forestry-related disturbances

Overall, patterns attributed to forestry-related disturbances were characterized by events that were 48 ha in size at the median and 261 ha on average (Table 3.3). Approximately 18% of the

Table 3.2 Summary of fire historical range-of-variability disturbance patterns.

	Range of variability			
Metric	x_{min}	$\tilde{x}$	x_{max}	$\bar{x}$ **(sd)**
Event area (ha)	11	589	28,040	2,159 (4,732)
Disturbed patch area (ha)	9	551	27,310	2,001 (4,538)
Island remnants area (ha)	0	78	23,774	758 (2,748)
Matrix remnants area (ha)	0	41	1,384	128 (226.4)
Total remnants area (ha)	0	133	24,450	887 (2,862)
Largest disturbed patch (LDPA) (ha)	9	491	27,267	1,941 (4,511)
Largest island remnant area (LIRA) (ha)	0	28	22,634	459 (2,439)
Mean island remnant area (MIRA) (ha)	0	9	354	32 (67)
Mean disturbed patch area (MDPA) (ha)	9	247	12,005	762 (1,563)
Number of island remnants	0	7	272	20 (38)
Number of disturbed patches	1	1	22	3 (4)
Percentage of disturbed patches (%DP)	70%	89	99	88.5 (6)
Percentage of island remnants (%IR)	0%	22	85	26 (20)
Percentage of matrix remnants (%MR)	0%	6	26	7 (6)
Percentage of total remnants (%TR)	0%	30	88	33 (21)
Percentage of largest disturbed patch (%LDP)	45%	88	99	85 (12)

Table 3.3 Summary table of comparisons between observed cumulative distribution functions (anthropogenic disturbances) and the empirical distribution function derived from the historical range-of-variability (HRV).

		Range of variability				MWW test[a]		K-S test[a]	
Disturbance Source	**Metric (*x*)**	x_{min}	$\tilde{x}$	x_{max}	$\bar{x}$ **(sd)**	***U***	**p-value**	***D***	**p-value**
Forestry	Event area (ha)	3	48	6,116	261 (817)	[b]	-	[b]	-
(*n* = 126)	Disturbed patch area (ha)	2	37	5,632	198 (663)	[b]	-	[b]	-
	Island remnants area (ha)	0	2	2,647	71 (284)	[b]	-	[b]	-
	Matrix remnants area (ha)	< 1	8	1,758	63 (194)	[b]	-	[b]	-
	Total remnants area (ha)	< 1	18	4,405	134 (457)	[b]	-	[b]	-
	Largest disturbed patch (LDPA) (ha)	2	28	3,514	131 (425)	[b]	-	[b]	-
	Largest island remnant area (LIRA) (ha)	0	2	911	25 (86)	[b]	-	[b]	-
	Mean island remnant area (MIRA) (ha)	0	1	55	6 (10)	[b]	-	[b]	-
	Mean disturbed patch area (MDPA) (ha)	2	19	302	33 (40)	[b]	-	[b]	-
	Number of island remnants	0	1	295	9 (33)	[b]	-	[b]	-
	Number of disturbed patches	1	2	50	4 (7)	[b]	-	[b]	-
	Percentage of disturbed patches (%DP)	51%	82	99	80 (12)	7,808	<0.001	0.39	<0.001*
	Percentage of island remnants (%IR)	0%	3	99	24 (33)	6,851	0.002	0.43	<0.001*
	Percentage of matrix remnants (%MR)	< 1%	18	49	20 (12)	1,718	<0.001	0.58	<0.001*
	Percentage of total remnants (%TR)	< 1%	33	100	44 (32)	4,678	0.07	0.21	0.018*
	Percentage of largest disturbed patch (%LDP)	17%	68	99	66 (24)	8,006.5	<0.001*	0.44	<0.001*
Aggregate harvest	Event area (ha)	3	45	2,015	174 (387)	[b]	-	[b]	-
(*n* = 74)	Disturbed patch area (ha)	3	35	1,331	125 (263)	[b]	-	[b]	-
	Island remnants area (ha)	0	< 1	279	26 (58)	[b]	-	[b]	-
	Matrix remnants area (ha)	0	6	756	49 (131)	[b]	-	[b]	-
	Total remnants area (ha)	< 1	9	980	75 (175)	[b]	-	[b]	-
	Largest disturbed patch (LDPA) (ha)	3	26	867	84 (167)	[b]	-	[b]	-
	Largest island remnant area (LIRA) (ha)	0	< 1	159	15 (33)	[b]	-	[b]	-
	Mean island remnant area (MIRA) (ha)	0	< 1	65	5 (12)	[b]	-	[b]	-
	Mean disturbed patch area (MDPA) (ha)	3	15	177	31 (36)	[b]	-	[b]	-
	Number of island remnants	0	1	72	5 (12)	[b]	-	[b]	-
	Number of disturbed patches	1	1	21	3 (4)	[b]	-	[b]	-
	Percentage of disturbed patches (%DP)	55%	84	100	82 (11)	4,432.5	<0.001*	0.35	<0.001*
	Percentage of island remnants (%IR)	0%	2	100	17 (31)	4,775	<0.001	0.56	<0.001*
	Percentage of matrix remnants (%MR)	0%	17	45	18 (11)	1,073.5	<0.001	0.59	<0.001*
	Percentage of total remnants (%TR)	2%	25	100	35 (29)	3,386	0.572	0.16	0.281*
	Percentage of largest disturbed patch (%LDP)	24%	82	100	71 (23)	4,370	<0.001	0.35	<0.001*

Table 3.3 (continued) Summary table of comparisons between observed cumulative distribution functions (anthropogenic disturbances) and the empirical distribution function derived from the historical range-of-variability (HRV).

		Range of variability				MWW test[a]		K-S test[a]	
Disturbance Source	**Metric (x)**	x_{min}	$\tilde{x}$	x_{max}	$\bar{x}$ **(sd)**	U	**p-value**	D	**p-value**
Dispersed harvest	Event area (ha)	3	48	4,901	159 (509)	[b]	-	-	-
(n = 98)	Disturbed patch area (ha)	2	43	2,779	106 (290)	[b]	-	-	-
	Island remnants area (ha)	0	10	633	45 (92)	[b]	-	-	-
	Matrix remnants area (ha)	< 1	9	2,122	53 (221)	[b]	-	-	-
	Total remnants area (ha)	< 1	25	2,755	98 (300)	[b]	-	-	-
	Largest disturbed patch (LDPA) (ha)	2	31	793	51 (85)	[b]	-	-	-
	Largest island remnant area (LIRA) (ha)	0	18	174	28 (35)	[b]	-	-	-
	Mean island remnant area (MIRA) (ha)	0	9	128	18 (24)	[b]	-	-	-
	Mean disturbed patch area (MDPA) (ha)	1	13	102	19 (17)	[b]	-	-	-
	Number of island remnants	0	2	136	5 (15)	[b]	-	-	-
	Number of disturbed patches	1	2	54	4 (6)	[b]	-	-	-
	Percentage of disturbed patches (%DP)	47%	82	99	80 (12)	6,084.5	<0.001*	0.42	<0.001*
	Percentage of island remnants (%IR)	0%	21	97	35 (36)	4,259	0.992	0.29	0.001*
	Percentage of matrix remnants (%MR)	1%	18	53	20 (12)	1,370	<0.001*	0.61	<0.001*
	Percentage of total remnants (%TR)	2%	55	100	54 (35)	2,918	<0.001	0.38	<0.001*
	Percentage of largest disturbed patch (%LDP)	8%	61	99	62 (26)	6,313	<0.001	0.49	<0.001*
Energy	Event area (ha)	< 1	1	491	2 (19)	[b]	-	-	-
(n = 1,057)	Disturbed patch area (ha)	< 1	1	451	2 (16)	[b]	-	-	-
	Island remnants area (ha)	0	0	332	1 (11)	[b]	-	-	-
	Matrix remnants area (ha)	0	0	126	< 1 (4)	[b]	-	-	-
	Total remnants area (ha)	0	0	372	1 (14)	[b]	-	-	-
	Largest disturbed patch (LDPA) (ha)	< 1	1	409	2 (14)	[b]	-	-	-
	Largest island remnant area (LIRA) (ha)	0	0	9	0 (28)	[b]	-	-	-
	Mean island remnant area (MIRA) (ha)	0	0	< 1	0 (2)	[b]	-	-	-
	Mean disturbed patch area (MDPA) (ha)	< 1	1	41	1 (1)	[b]	-	-	-
	Number of island remnants	0	0	1,239	2 (40)	[b]	-	-	-
	Number of disturbed patches	1	1	55	1 (2)	[b]	-	-	-
	Percentage of disturbed patches (%DP)	23%	100	100	98 (7)	6,631.5	<0.001*	0.83	<0.001*
	Percentage of island remnants (%IR)	0%	0	68	< 1 (3)	88,423	<0.001	0.92	<0.001*
	Percentage of matrix remnants (%MR)	0%	0	77	2 (7)	77,436.5	<0.001*	0.70	<0.001*
	Percentage of total remnants (%TR)	0%	0	77	2 (8)	86,852	<0.001	0.83	<0.001*
	Percentage of largest disturbed patch (%LDP)	3%	100	100	93 (17)	10,671.5	<0.001*	0.81	<0.001*

[a] Mann-Whitney-Wilcoxon (MWW) and Kolmogorov-Smirnov (K-S) tests were performed against the HRV (see Table 3.2).
[b] No tests were performed on absolute, area-based metrics.
* Indicates that ties were present in the data, therefore the calculated p-value is approximate.

median event area for these disturbances was represented by matrix remnants or ~12% higher than HRV observations (U = 1,718, $p < 0.001$; D = 0.58, $p < 0.001$) and only 3% was represented by island remnants or ~19% lower than HRV observations (U = 6,851, $p = 0.002$; D = 0.43, $p < 0.001$). The median %LDP was 68%, which was ~20% lower than HRV observations (U = 8,006.5, $p < 0.001$; D = 0.44, $p < 0.001$). MIRA for all forestry-related disturbances was 6 ha on average and only 1 ha at the median, but with zero-island events omitted the median adjusts to 5 ha.

3.3.4 Aggregate harvest designs

Aggregate harvest events were composed of few disturbed patches, one at the median and three on average (Table 3.3). %LDP accounted for 82% of the disturbance event area at the median, which was ~6% lower than HRV observations (U = 4,370, $p < 0.001$; D = 0.35, $p < 0.001$), and ranged in size from 3 ha at the minimum to 867 at the maximum. Island (%IR) and matrix remnants (%MR) represented 2% (U = 4,775, $p < 0.001$; D = 0.56, $p < 0.001$) and 16% (U = 1,073.5, $p < 0.001$; D = 0.59, $p < 0.001$) of the event area at the median, respectively, and were ~20% lower and ~10% higher than HRV observations, respectively. Total remnants (%TR), that is, matrix and island remnants combined, accounted for 25% of event area at the median and 35% on average, which was ~5% lower than HRV observations and was not significantly different from the HRV (U = 3,386, $p = 0.572$; D = 0.16, $p = 0.281$). MIRA for aggregate harvests was small, less than 1 ha at the median, but with zero-island events omitted the median adjusts to 5 ha, and only 5 ha on average. There were few island remnants, one at the median and five on average, despite these aggregate harvest designs reaching sizes of 45 ha at the median and 174 ha on average.

3.3.5 Dispersed harvest designs

%IR in dispersed harvest designs was 21% at the median, which was ~1% lower than HRV observations and was not significantly different from the HRV (U = 4,259, $p = 0.99$) and 34% on average, but the overall distribution was significantly different from the HRV (D = 0.29, p =

0.001). MIRA was quite large, 9 ha at the median and 18 ha on average. There were few disturbed patches in these harvest designs, two at the median and four on average. As a percentage, LDPA represented 61% of the total disturbance event area at the median, which was ~27% lower than HRV observations and significantly different from the HRV (U = 6,313, $p < 0.001$) and 62% on average. There were also few island remnants in dispersed harvests, two at the median and five on average. The largest island remnant in these harvest designs was 31 ha in size at the median and 51 ha on average. %MR represented 18% of total event area at the median, which was ~12% higher than HRV observations, and 20% on average. %TR accounted for 55% of total event area at the median, which was ~25% higher than HRV observations and significantly different from the HRV (U = 2,918, $p < 0.001$).

3.3.6 Energy-related disturbances

A high degree of variability was associated with energy-related disturbance metrics. For example, the number of island remnants ranged from zero to 1,239, zero at the median and two on average with a standard deviation of ±40. Although the presence of a dense seismic line network in the largest energy-related event resulted in 1,239 individual island remnants, totaling 332 ha in area, the 98th percentile of energy-related disturbances contained no island remnants at all. Energy-related disturbances were significantly different from the HRV in terms of all the metrics that were investigated. %IR and %MR was 0% (U = 88,423, $p < 0.001$; D = 0.92, $p < 0.001$) and 0% (U = 77,436.5, $p < 0.001$; D = 0.7, $p < 0.001$) at the median, which was ~22% and ~6% lower than HRV observations, and < 1% and 2% on average, respectively. %TR accounted for 0% of total event area at the median, which was ~30% lower than HRV observations (U = 86,852, $p < 0.001$; D = 0.83, $p < 0.001$), and 2% on average. By contrast, LDPA represented 100% of total event area at the median, which was 12% higher than HRV observations (U = 10,671.5, $p < 0.001$; D = 0.81, $p < 0.001$), and 93% on average.

3.4 Discussion

The areal pattern metrics are valuable in that they allow comparisons between different CRV sources, but they are less reliable as indicators of fire pattern emulation because they all co-vary with event size. Furthermore, since the 87 fires used for the HRV estimate were not chosen randomly, their sizes do not necessarily reflect an empirical distribution of fire size. The range of variability was recorded for all areal pattern metrics, but comparison tests to the HRV were not performed. However, the results may still be compared to other regional studies of empirical fire size. For example, (Cumming, 2001a) estimated the 95% confidence interval for maximum fire size in the study area to be between 227,000 ha and 1,820,000 ha, but the largest anthropogenic disturbance recorded was a 6,116 ha forest harvest. Additionally, the largest 1% of fires (> 1,000 ha) accounted for 98% of the total area burned/disturbed (Cumming, 2001a), but the largest 1% of forestry-related disturbances accounted for only 37% of the total area disturbed by forest harvesting. In other words, the majority of the landscape disturbed by forest harvesting is derived from many small harvest events. This finding supports other work where landscape patch size is considered to be outside the HRV due to forest harvesting and land management (Nonaka and Spies, 2005). The five percentage pattern measures are fire size-invariant (Andison, 2012), and thus form the basis for the CRV-HRV comparisons communicated here.

3.4.1 Disturbance patterns of the forestry and energy sectors

Forestry-related disturbances better approximated the HRV than did the energy-related disturbances in several important ways. Energy-related disturbances created a large number of extremely small, dispersed disturbances with virtually no surviving vegetation. In sharp contrast, the percentage of total remnants for forestry-related disturbances aligns very well to the HRV. Similarly, at the median, the largest disturbed patch for forestry was also fairly similar to that of the HRV (83% and 88%, respectively), compared to 100% at the median for energy-related disturbances. The high LDPA for energy-related disturbances reflects the fact that most (90%) of the disturbance events by the energy sector included only a single disturbed patch. Finally, although not statistically comparable to the HRV, it is notable that the median disturbance size of forest harvests was 48 ha, compared to less than 1 ha for energy-related disturbances. The data support the hypothesis that, relative to forest management, the energy sector creates a large number of very small evenly-distributed disturbance events with little or no residual vegetation.

The most notable trend observed from the analysis was the convergence of the percentage of total remnants (%TR) for forestry-related disturbances with the HRV. The fact that %IR was under-represented and %MR was over-represented relative to the HRV is of less concern, but signifies a sampling artifact whereby uncut areas within dispersed harvests were coded as matrix remnants. The physical distinction between island and matrix remnants is subtle, and in this case most of the islands were partially disturbed strip cuts. However, it is possible that these differences may relate to ecological function. Island remnants are physically isolated from the nearby matrix of intact forest and thus serve as refugia for small-bodied wildlife and seed sources for local flora (Banks *et al.*, 2011; Eberhart and Woodard, 1987; Eberhart, 1986). Matrix remnants are undisturbed forest residuals of disturbances that are physically attached to the surrounding matrix of intact forest, and function as corridors and cover for the movement of wildlife between forested patches (Renjifo, 2001).

The higher representation of %MR in forestry-related disturbances is an indication of fragmentation and is likely an artifact of dispersed harvesting strategies whereby cut blocks are spaced evenly throughout a landscape, thereby resulting in high levels of matrix remnants between cut blocks. However, many of the dispersed harvests that were sampled had only been through one of the two passes. If harvests that had been through two passes were sampled, then matrix remnants would likely be less represented. Although given this limitation, only a handful of the pattern indices quantified for dispersed harvests would be expected to deviate markedly from the current calculations if the second pass had been included. For example, island remnant levels, largest disturbed patch and mean patch size should not vary greatly from current quantities because this harvesting strategy reduces variability in patch size, but matrix remnant levels, total remnant levels, the number of disturbed patches and all of the areal metrics would certainly be affected. Furthermore, one challenge associated with sampling anthropogenic disturbances that vary across time, in addition to space, is assessing when an event is “complete” or discrete in time. This creates a special challenge for comparing dispersed harvests, which can take decades to “finish”, with fires that may burn over the course of weeks to months.

3.4.2 Disturbance designs of forest management

The analyses suggest that aggregate harvest patterns in the Al-Pac FMA better approximated the HRV than dispersed harvests in terms of event area, MDPA, disturbed patch density, %MR, %TR, and LDPA (Table 3.3). However, some dispersed harvest patterns were noteworthy in relation to fire patterns. For example, dispersed harvests generally better approximated the HRV in terms of absolute median sizes of spatial elements such as island, matrix and total remnants area, LDPA, LIRA, MDPA, and mean island area. Additionally, these disturbances featured more island remnants than aggregate harvests, but were also characterized by a higher percentage area represented by matrix remnants. Some of these differences can be explained by the fact that many of the dispersed harvests that were sampled had only been through one of the two harvest passes; after only one pass, the area designated for the second pass is matrix remnants. Thus what was captured in the results is a comparison of the current condition of disturbance patterns. As second passes are completed, many of the metrics for the dispersed harvesting pattern will shift. For example, if two passes are completed over a relatively short period of time, one would expect LDPA to increase and matrix remnants to decrease. However, extended periods between passes would effectively create two overlapping events, both with patterns that have no historical analogue. Overall, the effort to emulate fire disturbance patterns over the last several years has been successful, although further improvement is possible.

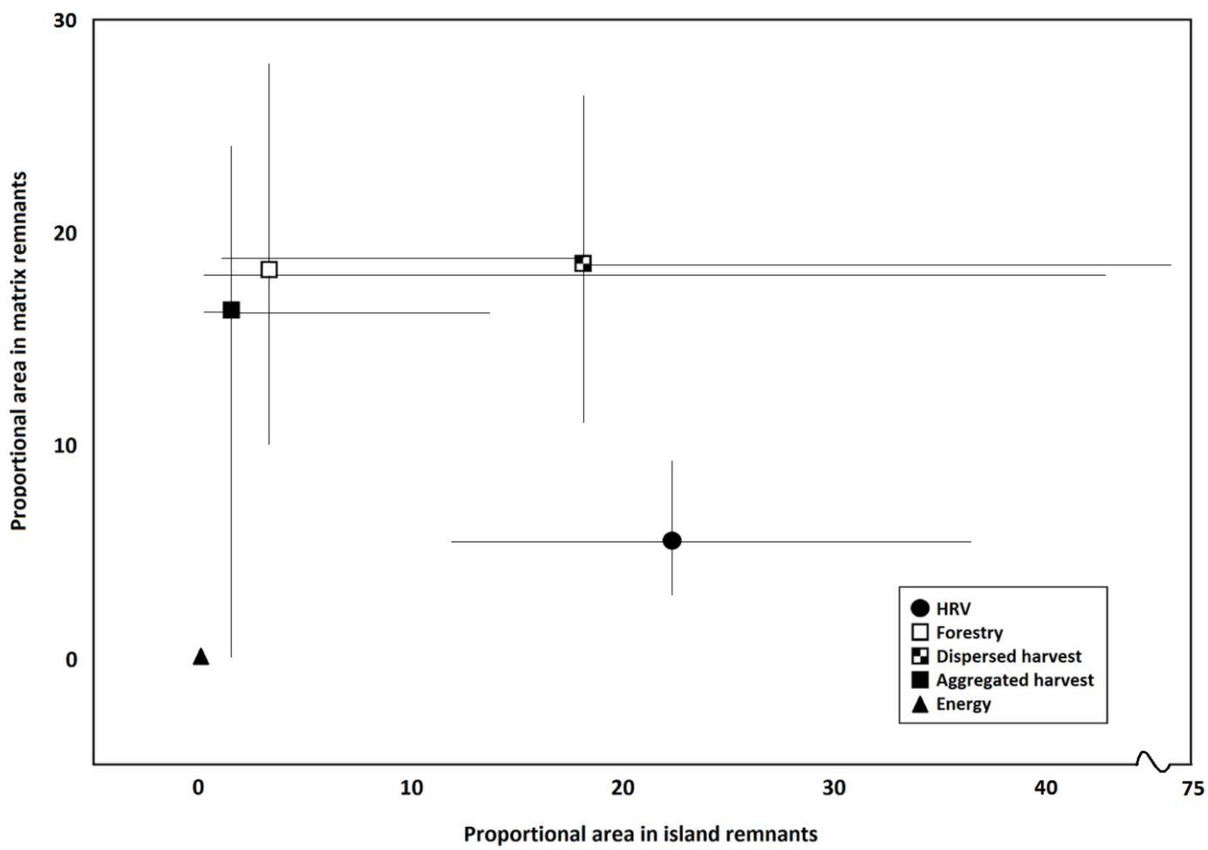
Although aggregate harvests did not perform better than dispersed harvests on absolute area-based metrics (*e.g.*, event area, island remnant area, LDPA, *etc.*), they did converge on historical targets for several key percentage metrics. Percentage metrics of the disturbance event area such as %LDP, %MR and %IR are critical for approximating historical mortality patterns, which are important post-disturbance habitats (Eberhart and Woodard, 1987). The lower percentage area occurring as matrix remnants in aggregate harvests is an indication that this harvesting strategy is achieving the goal of better approximating fire patterns on the landscape (Figure 3.4). The lower percentage of matrix remnants suggests that more of the disturbance area is allocated to disturbed patches thus resulting in a more compact event which better approximates historical levels and is more likely to reduce road density, and thus edge density, in harvest areas. Similarly, %LDP in aggregate harvests tended to account for a greater percentage of the disturbance event area than dispersed harvests, which is an important characteristic of historical fires. The sizes and percentage of island remnants in aggregate harvests diverged from historical levels more so than dispersed harvests and forest harvesting overall (Figure 3.4). However, most of the islands in dispersed harvest designs were partially disturbed strip cuts, many of which are designed for removal during a second pass. The fact that dispersed designs created larger disturbance events, combined with the relatively low area in disturbed patches, demonstrates one of the concerns of these harvesting strategies: vast harvesting "footprints" on the landscape. Aggregated harvest designs are more compact (*i.e.*, lower percentage of matrix remnants) meaning that they are more likely to leave significant areas of undisturbed forest in which no roads or management activities are required to be maintained for extended periods of time. Over time, one would expect that an aggregate harvesting strategy would result in fewer and larger forested patches across a given landscape; a pattern that is consistent with the historical fire regime.

There are some data artifacts worth noting for all disturbance sources. For example, %LDP is artificially inflated when there is only one disturbed patch per disturbance event, which was true of 90% of all energy-related disturbances. The tendency of dispersed harvests to approximate historical levels of island remnant-related metrics may be attributed to the higher likelihood of roads occurring in these events. Although the density of roads was not quantified or analyzed in this study, the effects of roads on the generation of spatial elements like island remnants is related to the spatial language itself.

3.4.3 Management interpretations

In order for ecosystem-based initiatives to be successful as coarse-filter approaches to biodiversity conservation, management strategies must be implemented that approximate the range-of-variability of historical disturbance patterns. For forest planners, the sizes and percentage of island and matrix remnants remains a critical characteristic of forest disturbances which is important for maintaining core and refugia habitats for a variety of forest mammals and avifauna (Eberhart and Woodard, 1987; Eberhart, 1986; Matthiae and Stearns, 1981; Whitcomb *et al.*, 1981). Although forestry-related disturbances generated less island remnants, and more

Figure 3.4 Plot of median island and matrix remnants as a percentage of total event area by disturbance class. The whiskers represent the upper bound 75th quartile and the lower bound 25th quartile for both axes.



matrix remnants compared to the HRV, combined total remnants were not significantly different from the HRV at the median.

The physical distinction between island and matrix remnants is subtle. However, it is possible that these differences may relate to ecological function. Island remnants are physically isolated from the nearby matrix of intact forest and thus serve as refugia for small-bodied fauna and seed sources for local flora (Banks *et al.*, 2011; Eberhart and Woodard, 1987; Eberhart, 1986). Matrix remnants are undisturbed forest residuals of disturbances that are physically attached to the surrounding matrix of intact forest, and functions as corridors and cover for the movement of fauna between forested patches (Renjifo, 2001).

Historical remnant patterns may also mitigate concerns about larger anthropogenic disturbance events. For example, the inclusion of matrix remnants may allow greater flexibility within larger harvests that may be beneficial for larger-bodied mammals, which prefer the dense understory foliage of forests for foraging in the fall (Lewis, 2002). Yet, larger harvest areas alone do not necessarily replicate the heterogeneity of post-burn, structural characteristics of historical forested systems. Thus, disturbance patterns in isolation of other factors like biotic feedbacks which affect the efficacy of succession-based ecosystem restoration strategies (Song, 2002; Suding *et al.*, 2004). For example, there is no provincial mandate for the energy sector to re-vegetate seismic lines or well sites in Alberta and native species re-vegetation is costly and voluntary (MacFarlane, 2003, 1999).

The disturbance event scale of analysis presented here has some disadvantages for planning at the landscape scale. For example, at the resolution of the landscape, non-harvested forest matrix between disturbances and matrix remnants between incomplete multi-pass harvests provide cover and function as corridors for fauna, but these elements of landscape composition are not captured by the results. Thus, comparison of HRV and anthropogenic patterns at larger spatial scales is an area requiring future research. The cumulative nature of anthropogenic disturbances on the Alberta landscape was not considered by this study. However, investigations into the sources of and interactions between anthropogenic disturbances would greatly benefit the cooperative land planning initiatives between the energy and forestry sectors in Alberta. Furthermore, disturbance-based forest research would benefit from future lines of inquiry that examine the cumulative nature of anthropogenic disturbance activities. Such research should not only investigate the combined pattern impacts of forestry and the energy sector, but also the nature and spatial configuration of linear right-of-ways such as roads, seismic lines, pipelines and transmission lines. These linear features are ubiquitous in Alberta in particular due to rich energy deposits and function as potential sources of both fragmentation and dispersal networks for mammals, forest avifauna and flora (MacFarlane, 2003; Machtans, 2006). Ideally, such studies would be augmented with fine-filter research and effectiveness monitoring if EBM approaches are actually fulfilling the biotic response assumptions inherent in their principles.

The historical role of First Nations burning in the central mixedwoods of Alberta is underrepresented in historical ignition data. First Nations burning in northern Alberta tended to be seasonal in order to manipulate the distribution of regional fauna and flora (Lewis, 1982). The degree to which historical First Nations burning altered landscape patterns in the northern Boreal Plains has been documented because fire ignitions by the Plains Cree occurred well into the 1940s in northern Alberta (Lewis and Ferguson, 1988). Few studies incorporate these pre-industrial anthropogenic disturbance activities into account when considering local flora and fauna, yet the effects of such anthropogenic influences on the spatiotemporal distribution of understory vegetation remains an important factor in the consideration of faunal foraging, habitation and movement throughout boreal landscapes (Lewis, 1982).

Finally, the elapsed time between dispersed harvest passes is crucial for modeling these disturbances. Many of the disturbances that are communicated here were incomplete at the planning level due to a multiple-pass design. In addition, the road network associated with dispersed harvesting can fragment a landscape for several decades. Completing all harvests passes as near together in time would better emulate historical fires and reduce the required road infrastructure.

Chapter 4

The spatial patterns of anthropogenic disturbance in the western Canadian boreal forest following oil and gas development

4.1 Introduction

The boreal forest is one of the planet's largest and most important biomes. In Canada, the boreal zone spans 552 Mha, of which more than half is forested and considered intact (Brandt, 2009; Lee *et al.*, 2003). The boreal forest provides habitat for many species and plays a critical role in provisioning ecosystem services such as filtering water, storing carbon, and regulating climate (Millennium Ecosystem Assessment, 2005).

Human appropriation of the boreal forest has been ocurring for millennia and the anthropogenic disturbance regime has transitioned several times (Timoney, 2003). Prior to European settlement, First Nations routinely set localized fire to boreal forests to promote re-growth, clear trails and attract game fauna (Lewis, 1982). Since industrialization, the majority of the southern extents of the boreal forest has been appropriated for wood fiber production or has been converted for agriculture (Delong and Tanner, 1996; Hobson *et al.*, 2002; Timoney, 2003). Today, rates of anthropogenic disturbance exceed all other disturbance sources in some regions of the boreal forest associated with intensive forest management (Pickell *et al.*, 2014b; Schneider *et al.*, 2003). The cumulative area of anthropogenic disturbance in the boreal forest *circa* 2010 was estimated at 23 Mha (Pasher *et al.*, 2013).

Oil and gas development has resulted in significant anthropogenic disturbance in much of the western Canadian boreal forest over the last 20 years (Lee and Boutin, 2006; Lee and Cheng, 2009; Timoney and Lee, 2001). Hydrocarbons such as natural gas, coal and bitumen underlay forest ecosystems throughout the western Canadian boreal forest and a great effort has been made in recent years to extract these resources since the price for energy has risen considerably (Lee and Cheng, 2009). Many of these oil and gas deposits were historically too deep to recover, but technological innovations have significantly increased the accessibility of these resources in recent years (Rogner, 1997). Horizontal and directional drilling, hydraulic fracturing, and steam-assisted gravity drainage have significantly improved *in situ* extraction capabilities of hydrocarbons (Butler, 1985). These innovations have resulted in significant oil and gas development in Alberta, where approximately 14,000 well sites were drilled every year over the previous decade for *in situ* production (Alberta ESRD, n.d.). Despite the small size of well site disturbances (~1 ha), the ubiquitous distribution of these features on the Alberta landscape contributes to a much larger areal footprint (Pasher *et al.*, 2013), and reclaiming these sites to native forest has proved to be challenging in the boreal (MacFarlane, 1999; Osko and Macfarlane, 2001). Oil and gas development has been associated with biodiversity loss (Butt *et al.*, 2013; McDaniel and Borton, 2002), attenuated habitat use and habitat loss for a wide range of taxa (Northrup and Wittemyer, 2013), and reduced delivery of ecosystem services (Pickell *et al.*, 2014a).

Anthropogenic disturbance often generates forest patterns that have no historical analogue in the boreal forest (Pickell *et al.*, 2013; Schneider *et al.*, 2003). Industrial access corridors such as roads and cut lines are characteristically distinct from other disturbances such as forest harvesting in that they are generally linear, which both promotes and inhibits the movement and dispersal of organisms (Lesmerises *et al.*, 2012). These features are highly persistent due to repeat anthropogenic use and permanent land conversion (Lee and Boutin, 2006). For these reasons, multiple-use landscapes in the boreal forest have undergone significant biological invasion (Frelich *et al.*, 2006), changes to vegetation composition and persistence (Hebblewhite *et al.*, 2005; Lee and Boutin, 2006), and habitat conversion (Dyer *et al.*, 2001; Northrup and Wittemyer, 2013). Quantifying changes to disturbance patterns relative to an historical benchmark may provide insights into the mechanisms driving biodiversity and habitat availability in multiple-use landscapes.

Historical range-of-variability (HRV) analysis is well-suited for cumulative impacts monitoring in multiple-use landscapes. The HRV concept is predicated on the idea that the biota of a given locale have adapted to respond to a range of conditions that permit a shifting mosaic of multiple states to persist given varying degrees of disturbance (Delong, 2002; Hunter, 1993; Landres *et al.*, 1999). For example, disturbance frequency controls the amount of old forest (Andison and Marshall, 1999), disturbance extent is important for post-disturbance floristic composition (Mayor *et al.*, 2012), and the biological legacies of fire create refugia for nonmobile species, provide habitat, forage, and cover, and assist in re-populating disturbed areas (Robinson *et al.*, 2013).

An emerging concern in the western Canadian boreal forest is that spatial patterns of anthropogenic disturbance are outside the HRV. More than 20 years ago, Hunter (1993) called for forestry activities to emulate the spatial characteristics of historical disturbance regimes. Since then, several studies have examined the efficacy of such forestry-based initiatives with results that indicate scale- or regional-dependence. At the landscape scale, Pickell *et al.* (2013) found that HRV-based harvest designs were significantly more similar to fire than the traditional multi-pass harvesting system in northern Alberta. Andison and Marshall (1999) found that HRV-based guidelines for patch size and relative frequency of seral stages for a landscape in British Columbia did create patterns marginally closer to HRV targets than traditional harvesting practices. However, there has been little research on whether cumulative anthropogenic disturbance patterns from multiple land uses are outside the HRV in the western Canadian boreal forest (Schneider *et al.*, 2003). More specifically, it is unknown what the contribution of energy sector disturbance has been on boreal forests in recent decades.

The research presented in this chapter examines managed landscapes in two western Canadian boreal forests impacted by oil and gas development. A temporal analysis of the last 60 years of anthropogenic disturbance was undertaken to assess whether cumulative anthropogenic disturbance patterns vary from an HRV of fire burn patterns. The analysis was driven by three overarching questions. Are anthropogenic disturbance patterns within the HRV? Do anthropogenic disturbance patterns vary by region and management history? Have cumulative anthropogenic disturbance patterns changed through time in response to oil and gas development? To answer these questions, I will: (1) map historical anthropogenic disturbances for two boreal forest landscapes using aerially-interpreted forest inventory data; (2) evaluate how anthropogenic disturbance patterns vary regionally and from an HRV of fire burn patterns; and (3) evaluate how anthropogenic disturbance patterns vary between two distinct resource extraction periods.

4.2 Materials and methods

4.2.1 Study areas

I sampled anthropogenic disturbance patterns in two ecologically-distinct regions of the western Canadian boreal forest where significant oil and gas development has occurred over the last two decades and with significant differences in fire regime characteristics: the Boreal Forest and the Foothills natural sub-regions of Alberta (Natural Regions Committee, 2006). Both sub-regions are part of the Boreal Plain ecozone according to the Canadian forest ecosystem classification system. The Boreal Forest sub-region (hereafter Boreal Forest) is a regional ecological system with mostly flat undulating terrain (Beckingham and Archibald, 1996). The Boreal Forest is deciduous-leading mixedwoods comprised primarily of *Populus*, *Larix*, and *Picea* species (Natural Regions Committee, 2006). The Foothills sub-region (hereafter Foothills) is a provincial subdivision of the Montane Cordillera and Boreal Plain ecozones. The topography of the Foothills is highly variable, with elevations ranging from 500 m.a.s.l. to 1,500 m.a.s.l. (Beckingham *et al.*, 1996). The Foothills is dominated by conifer-leading species such as *Pinus* and *Picea* species (Natural Regions Committee, 2006).

Both sub-regions are rich with hydrocarbon deposits. Bitumen occurs throughout the Athabasca drainage basin in the Boreal Forest and is primarily mined near Fort McMurray, Alberta. In other areas, *in situ* projects recover deeper bitumen deposits from well sites. Large natural gas deposits exist south of the mineable bituminous sands region, which extends west toward the Rocky Mountain foothills. In the Foothills, most oil and gas activity is related to natural gas extraction and coal mining.

4.2.2 Disturbance sampling

Three forest management zones in Alberta were selected for study due to the history of oil and gas development on the landscapes. Seven equally-sized sample areas were located within the forest management zones, which were approximately proportional to the forested area under management. Four sample areas were located in the Alberta-Pacific Forest Industries (Al-Pac) forest management area (FMA) in the Boreal Forest. Two sample areas were located in the Hinton Wood Products (HWP) FMA, and one sample area was located in the adjacent Alberta

Newsprint Company (ANC) FMA in the Foothills (Figure 4.1). The locations of the sample areas were selected to represent the range of anthropogenic disturbances within the forest management zones. For example, the sample areas in the Al-Pac FMA were initially selected for a range of forest harvesting practices and intensity of energy sector disturbance in order to be representative of the largest FMA in Alberta (Pickell *et al.*, 2013). The samples in the Foothills were also selected to represent various forestry practices, natural gas production, and coal mining across two of the oldest FMAs in Alberta. Each sample area was a square 100,000 ha in size, which allowed the range of anthropogenic disturbance patterns occurring at the landscape scale to be captured (Schneider *et al.*, 2003).

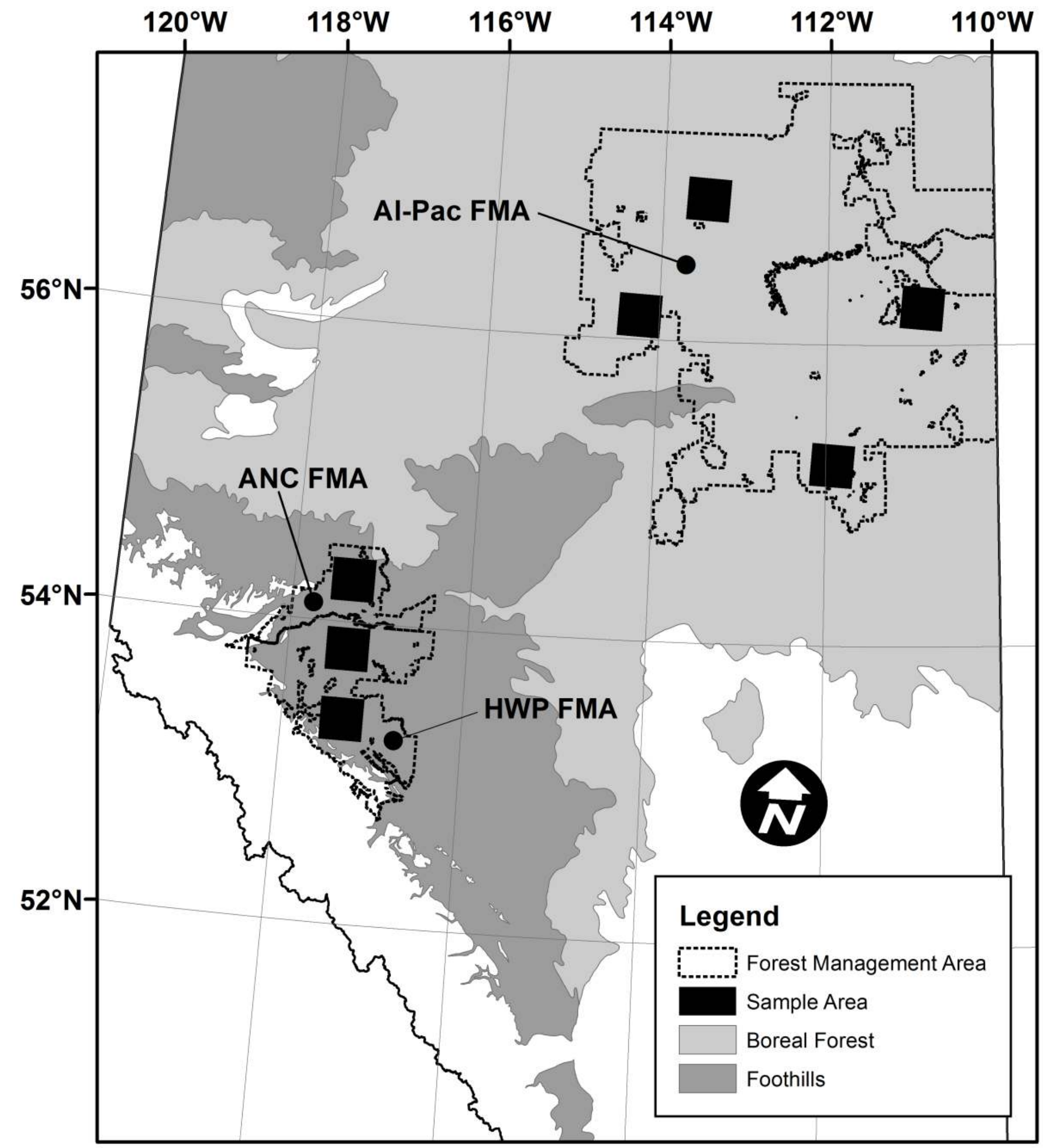
4.2.3 Overview of disturbance event mapping

Within each sample area, anthropogenic disturbances were mapped spatially and temporally using several remote sensing approaches that ranged in spatial scale. First, target features related to forest harvesting and energy development were extracted from digitized forest inventory data derived from interpreted aerial photographs. Most features were attributed with a year of disturbance, although many features related to energy development did not contain this information. For those features, I derived a year of disturbance from an annual time series of Landsat observations. For some harvest areas, residual trees were not mapped in the inventory data, so an ancillary light detection and ranging (LiDAR) dataset was used to map structures above 3 m. Finally, anthropogenic disturbance events were characterized using a pattern analysis tool that aggregates mapped features into disturbance events. The data and processes are described in further detail in the following sections.

4.2.4 Anthropogenic disturbance data

Aerially-interpreted Alberta Vegetation Inventory (AVI) data from 2010 in the Boreal Forest and 2012 in the Foothills were used to map anthropogenic forest crown mortality. These data were mapped from air photographs at 1:40,000 and 1:60,000 scales with a minimum polygon size of 2 ha for anthropogenic features (Alberta ESRD, 2005). The attributes for these data included the

Figure 4.1 Map of the study regions and location of the sample areas.



feature identity and year of disturbance. Features mapped included cut blocks, well sites, pipelines, gravel pits, surface mines, roads, and railroads. Supplementary datasets of seismic lines (*i.e.*, narrow linear forest clearings used for subterranean exploration of hydrocarbons), roads and transmission lines were obtained from the respective forest management companies that operate in the study areas. The seismic lines were buffered according to the width attribute in the data or to 2 m based on other studies conducted in the same region (Lee and Boutin 2006). Roads, transmission lines, railways, and pipelines were combined into a single right-of-way layer and buffered by 50 m. This wide buffer ensured that the right-of-way network was continuously connected and that all potential Geographic Information System (GIS) errors were minimized. Finally, the data were spatially generalized to minimize error from the mapping and interpretation process.

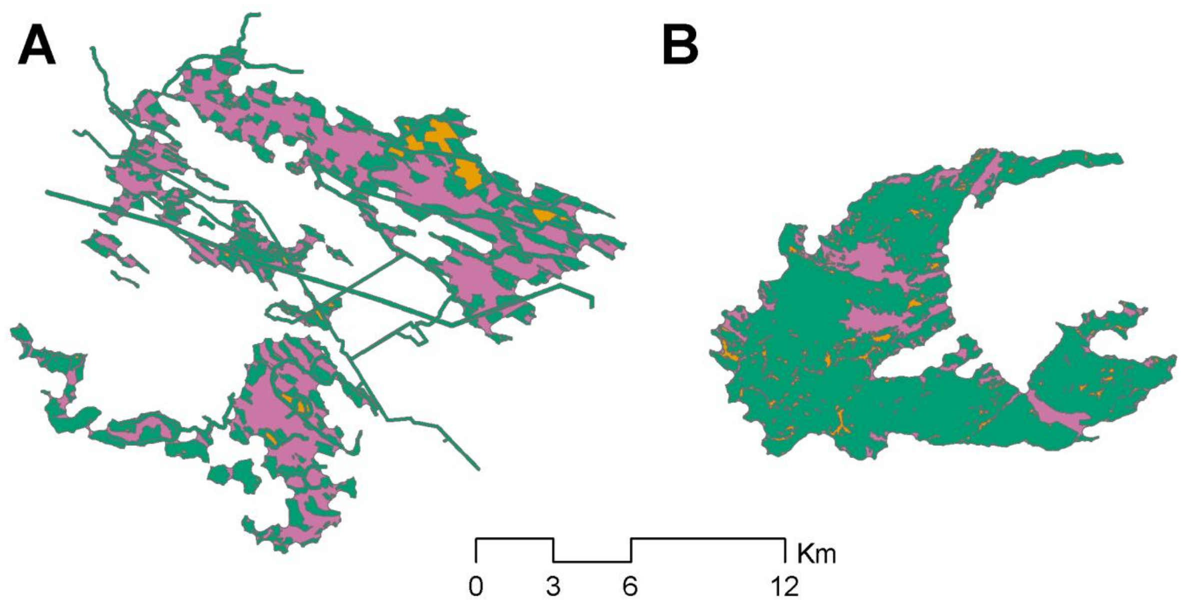
The AVI data were augmented to create standardized input layers across all sample areas. A disturbance year was not available for well sites within the Al-Pac study area. The vegetation change tracker (VCT) (Huang *et al.*, 2010) was used to estimate a year of disturbance from an annual Landsat time series stack (1984 to 2011), and attributed a disturbance year to the AVI well site polygons by pixel majority. A disturbance year was recovered for approximately 89% of the total well sites in the Al-Pac study area. The remaining 11% of mapped well sites were identified by the VCT as persistent non-forest (*i.e.*, cleared before 1984). These features were included in the analysis of the 1972 to 2001 period. This was justified since well sites were assumed to persist in a disturbed state for 40 years and the analyzed period was only 30 years long. Another data gap that was encountered was that the oldest island remnants were not mapped in the AVI data for harvest blocks on the HWP sampling areas. In this case, I used an ancillary LiDAR dataset acquired between 2005 and 2008 to retroactively map areas where crown heights were at least 3 m (the height assumed to be the undisturbed state in the model) and polygons were generated in a GIS with a minimum mapping unit of 2 ha, which was equivalent to the AVI data.

4.2.5 Characterizing disturbance event patterns relative to HRV

In order to assess whether anthropogenic disturbance patterns were outside the HRV, I compared the sampled anthropogenic disturbance patterns to fire disturbance patterns from a systematic analysis of western Canadian fires by Andison (2012) and Andison and McCleary (2014). In other words, the HRV represented a control for studying the change in anthropogenic disturbance patterns between the two periods and provided context for those analyses. Using air photographs and negatives, Andison (2012) interpreted the crown mortality levels of 129 fires with no record of suppression that spanned more than 100 Mha of the western Canadian boreal forest from 1939 to 2006. Andison and McCleary (2014) determined that fire disturbance patterns could be categorized into two regional disturbance regimes: one for the boreal forest and one for the boreal foothills, which correspond to the Boreal Forest and Foothills sub-regions, respectively (Figure 4.1). Since the fires were not sampled randomly with respect to size, further comparison with anthropogenic disturbance patterns were limited primarily to metrics standardized by event area.

The fire pattern results from Andison and McCleary (2014) are embedded within a web-based decision-support tool called NEPTUNE (Novel emulation Pattern Tool for Understanding Natural Events). NEPTUNE is comprised of a series of algorithms that aggregate mapped features identified in a GIS environment into spatially- and temporally-discrete disturbance events using buffering logic (Andison and McCleary, 2014; Andison, 2012; Pickell *et al.*, 2013). Mapped patches comprising each disturbance event can be resolved into four levels of mortality based on percent crown removal: fully disturbed (≥95%), partially-disturbed island remnant (<95%), intact island remnant (0%), and intact matrix remnant (0%). Supported input spatial layers include harvest blocks, roads, well sites, seismic lines (*i.e.*, cut lines), and permanent water features. Island remnants may be mapped *a priori* (*e.g.*, partial and selective harvests); otherwise NEPTUNE identifies the "holes" in the input disturbed features and re-classifies them as intact island remnants. Finally, matrix remnants are generated between all disturbed patches as an artifact of the spatial proximity and complexity of disturbed features (Figure 4.2). Matrix remnants were generated using a buffering algorithm based on a distance of 200 m, which Andison (2012) suggested was the optimum distance for amalgamating the patches of fires in the western Canadian boreal forest. The buffer distance was a compromise between estimating the

Figure 4.2 Examples of mapped disturbance events: forest harvesting and roads during the early period in the Foothills (A); and a reference fire that burned in 1956 in southern Alberta (B).



area of the disturbance event and grouping the disturbed patches into a singular, spatially-discrete disturbance event. Only well sites and cut blocks are used to determine the size and perimeter of anthropogenic disturbance events. Roads, seismic lines, and all other linear disturbances are used to ensure that disturbed area is properly accounted for, but are not used to define the boundary of disturbance events. However, linear disturbances do determine whether a patch is classified as a matrix remnant or island remnant. For example, an island remnant is detached from the surrounding intact forest matrix while a matrix remnant is the intact forest matrix fragment that occurs within the disturbance event perimeter (Figure 4.2). Finally, permanent water features are masked from the analysis and final outputs.

4.2.6 Pattern analysis assumptions

A spatial buffer and time interval threshold were set prior to performing any analyses in NEPTUNE. These options govern how the mapped features are aggregated into disturbance events in space and time, respectively. I selected a spatial threshold of 200 m to be consistent with previous research (Andison and McCleary, 2014; Andison, 2012; Pickell *et al.*, 2013). The time intervals were equivalent to the periods of major resource development that were selected under the advisement of local experts with knowledge of changes to resource management practices and are summarized in Table 4.1.

Each disturbance feature layer used in the NEPTUNE tool required an assumption of recovery time to at least 95% canopy cover. The recovery times used in NEPTUNE are summarized in Table 4.2. For all study areas, cut lines and well sites were assumed to persist for 40 years from the disturbance year identified in the dataset. For most of the industrial features related to energy extraction, 40 years extends beyond present day from the original year of disturbance, so it was assumed that these features never recovered in the model. This choice of recovery time for oil and gas features is consistent with previous research in the study area (Lee and Boutin, 2006; MacFarlane, 1999; Osko and Macfarlane, 2001; Van Rensen, 2014). The recovery time for forest harvests was set to 15 years based on the overall site conditions and the leading tree species of the study areas, using site index equations for northern Alberta Ecosites (Beckingham and Archibald, 1996; Beckingham *et al.*, 1996). Finally, roads

Table 4.1 Dates of the analyzed resource extraction periods by management area and the percent of anthropogenic disturbance area represented by oil and gas features (e.g., well sites and seismic lines).

Management area (region)	Resource extraction period (years)	Oil and gas as a percentage of anthropogenic disturbance
Hinton Wood Products (Foothills)	Early period: 1956 to 1999 (44)	22
	Late period: 2000 to 2012 (13)	78
Alberta Newsprint Company (Foothills)	Early period: 1949 to 1999 (51)	38
	Late period: 2000 to 2011 (12)	62
Alberta-Pacific (Boreal Forest)	Early period: 1972 to 2001 (30)	36
	Late period: 2002 to 2010 (9)	64

Table 4.2 Summary of disturbance levels and recovery intervals used to map anthropogenic disturbance.

Disturbance type	Recovery assumption to mature forest at 3m (years)	Reference
Harvest block	15	Beckingham and Archibald (1996), Beckingham *et al.*, (1996)
Well site	40	Osko and Macfarlane (2001)
Seismic line	40	Lee and Boutin (2006), MacFarlane (1999), Osko and Macfarlane (2001), Van Rensen (2014)

were assumed to recover 100 years after disturbance simply to ensure that all permanent right-of-ways remained as persistent features on the landscapes.

4.2.7 Statistical analysis of pattern metrics

The anthropogenic disturbance events were analyzed for 10 key pattern metrics (Table 4.3). Due to the fact that many areal metrics correlate with event area (Andison, 2012), the analysis was constrained to the percentage metrics when evaluating whether anthropogenic disturbance patterns varied from the HRV. Anthropogenic disturbances in the Foothills were compared to the Boreal Foothills fire patterns and anthropogenic disturbances in the Boreal Forest were compared to the Boreal Forest fire patterns. I also compared the sub-regions as well as the early and late periods. I choose the year 2000 as the threshold between periods because it roughly corresponds with the beginning of HRV-based forest management and significant increase in oil and gas development (Van Rensen, 2014) (Table 4.1).

Differences were assessed using non-parametric methods. The Kolmogorov-Smirnov (K-S) test was performed as a two-tailed test of deviation between the sample cumulative distribution function (CDF; *i.e.*, anthropogenic disturbance) and the control CDF (*i.e.*, fire disturbance) with $\alpha = 0.05$. The rank sum Mann-Whitney-Wilcoxon (MWW) test was performed as a two-tailed test of deviation in location (median) between the sample CDF and the control CDF. For the MWW test, all variables were $\log(x + 1)$-transformed to reduce heteroscedasticity. Significant differences identified from the MWW tests were further scrutinized *post hoc* by fitting 95% confidence bounds to the median and comparing the interquartile range of sample distributions using box-and-whisker plots.

4.3 Results

A total of 2,329 anthropogenic disturbance events were mapped comprising 141,115 ha of disturbed forest area (Table 5.4). These disturbance events were distributed among the study areas as follows: 919 (32,254 ha) in the four Al-Pac samples; 848 (80,746 ha) in the two HWP

Table 4.3 Analyzed pattern metrics of disturbance events and method of quantification. All metrics are calculated on a per event basis and are detectable to a resolution of 0.02 ha.

Pattern metric	Interpretation	Method of quantification (units)
Event area (EA)	The total area of influence of a disturbance that is discrete in space and time	Total remnants + disturbed area (ha)
Mean disturbed patch area (MDPA)	The average size of all disturbed features (*e.g.*, harvest block, well site)	Total island remnants + disturbed area / number of disturbed patches (ha)
Largest disturbed patch area (LDPA)	The largest size of all disturbed features of an event	Maximum disturbed patch area / event (ha)
Mean remnant island area (MIRA)	The average size of all island remnants	Total island remnant area / number of island remnants (ha)
Largest island remnant area (LIRA)	The largest size of all island remnants	Maximum island remnant area (ha)
Percentage area of island remnants (%IR)	The relative area of island remnants	Island remnants area / event area * 100 (%)
Percentage area of total remnants (%TR)	The relative area of island and matrix remnants (*i.e.*, total remnants)	Total remnants area / event area * 100 (%)
Percentage area of largest disturbed patch (%LDP)	The relative area of the largest disturbed patch	Largest disturbed patch area / event area * 100 (%)
Event edge density (ED)	The ratio of edge perimeter to event area	Edge length of harvests, seismic lines, roads, and well sites / event area ($km \cdot ha^{-1}$)

samples; and 562 (28,118 ha) in the ANC sample. Approximately 1,149 disturbance events (94,973 ha) occurred during the early period (between 1949 and 1999) and 1,180 disturbance events (46,142 ha) occurred during the late period (between 2000 and 2012).

4.3.1 Trends by disturbance event size class

The size class distribution of disturbance events was highly skewed. Small disturbances (< 3 ha) accounted for almost 70% of the number of events, although only 2% of the area disturbed (Table 4.4). In the Foothills, small disturbances accounted for 39% of the events during the early period and 70% during the late period (< 1% and 3% by area, respectively). As a rate standardized by time, this translated into 35 small disturbances per decade during the early period, compared to 547 small disturbances per decade during the late period. In the Boreal Forest, the relative density of small disturbances declined from 85% to 77% (4% to 1% by area). This equated to 204 and 169 small disturbances per decade during the early and late periods, respectively.

Several key pattern metrics varied considerably by disturbance size class and between sub-regions. In both sub-regions, the smallest disturbances were composed primarily of a single patch with the lowest levels of remnants of any size class (Figure 4.3). Generally, the greatest levels of island remnants (%IR) occurred in events between 100 and 1,000 ha in size and decreased between the early and late periods (Figure 4.3). Percentage area of largest disturbed patch (%LDP) increased between the early and late period for events between 100 and 1,000 ha in the Foothills (Figure 4.3). Percent area of total remnants (%TR) increased linearly with event size, and decreased between the early and late periods in both sub-regions (Figure 4.3).

4.3.2 Trends by number of patches per event

The number of disturbed patches and island remnants per event was highly skewed. Disturbance events were stratified into two classes comprised of single patch events (SPE) composed of only one disturbed feature (*i.e.*, one well site or one harvest block) and multiple patch events (MPE)

Table 4.4 Summary of the number and area of single-patch events (SPE) by event size class, region, and period.

Metric	Boreal Forest – Early Period						
	Event size class (ha)						
	0-3	3-10	10-100	100-1 000	1 000-10 000	> 10 000	Total
SPE (n)	600	10	29	1	0*	0	640
SPE area (ha)	600	56	848	192	0*	0	1,696
Total disturbed area (ha)	629	75	2,201	8,913	5,137	0	16,955
Total events (n)	613	15	60	31	2	0	721

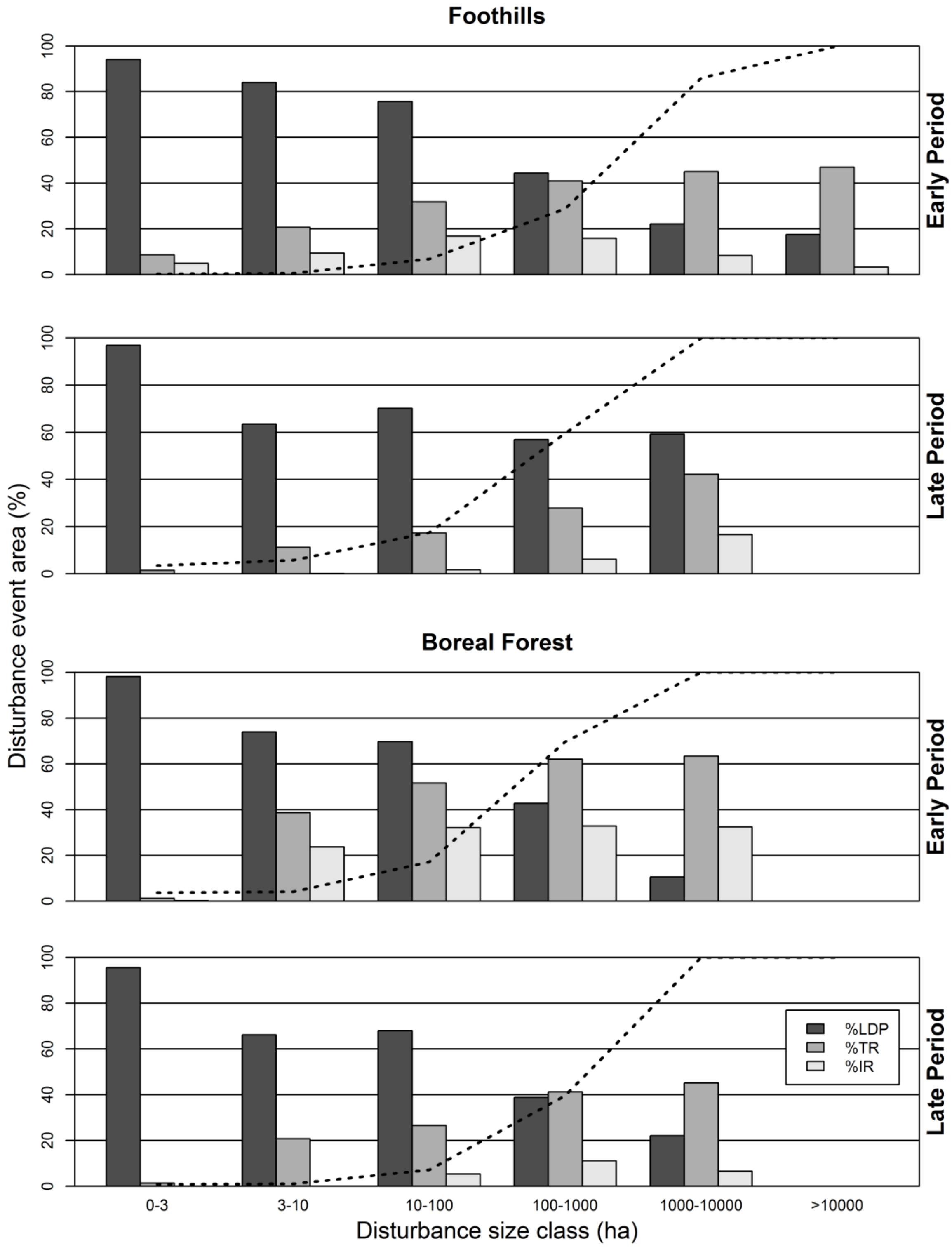
Metric	Boreal Forest – Late Period						
	Event size class (ha)						
	0-3	3-10	10-100	100-1 000	1 000-10 000	> 10 000	Total
SPE (n)	143	4	13	0*	0*	0	160
SPE area (ha)	114	28	425	0*	0*	0	567
Total disturbed area (ha)	129	33	936	4,961	9,239	0	15,298
Total events (n)	152	5	22	13	6	0	198

Metric	Foothills – Early Period						
	Event size class (ha)						
	0-3	3-10	10-100	100-1 000	1 000-10 000	> 10 000	Total
SPE (n)	155	45	75	7	0*	0*	282
SPE area (ha)	235	229	2,709	1,475	0*	0*	4,648
Total disturbed area (ha)	262	280	4,898	17,123	44,530	10,925	78,018
Total events (n)	169	54	130	61	13	1	428

Metric	Foothills – Late Period						
	Event size class (ha)						
	0-3	3-10	10-100	100-1 000	1 000-10 000	> 10 000	Total
SPE (n)	652	47	35	4	0*	0	738
SPE area (ha)	1,000	226	910	1,682	0*	0	3,818
Total disturbed area (ha)	1,075	723	3,617	12,972	12,457	0	30,844
Total events (n)	684	148	103	41	6	0	982

SPE = disturbance events comprised of one disturbed patch.
* all disturbance events had more than one patch.

Figure 4.3 Anthropogenic disturbances comprised by percent largest disturbed patch (%LDP), percent total remnants (%TR) and percent island remnants (%IR) by study region, period, and size class. Dashed line indicates the cumulative percent of disturbed area by size class.



as an initial aid for interpretation (Table 4.4). Approximately 80% of anthropogenic disturbance events were SPE; however SPE represented less than 8% of the total area disturbed (Table 4.4). In the Boreal Forest the relative abundance of SPE decreased from 89% during the early period to 81% during the late period. In the Foothills, the relative abundance of SPE increased from 66% during the early period to 75% during the late period. In both sub-regions, MPE had greater remnants levels, larger disturbed patches, and lower edge density than SPE (Table 4.5).

The number of island remnants per event was also highly skewed relative to disturbed area. In the Boreal Forest, 89% of all anthropogenic disturbances had zero island remnants; however, those events only accounted for approximately 6% of the total disturbed area. In the Foothills, 82% of events had zero island remnants and those events accounted for 7% of the total disturbed area.

4.3.3 Regional trends and comparisons to the HRV

Anthropogenic disturbance patterns were generally different from historical fire burning patterns (*i.e.*, HRV), although several notable trends varied by region. The most significant differences were related to disturbed patch size and levels of undisturbed remnants.

Disturbed patch size increased among anthropogenic disturbances in the Boreal Forest and decreased in the Foothills between the early and late periods (Figure 4.4). Mean disturbed patch area (MDPA) was about 33 ha smaller than the HRV in the Foothills and about 203 ha smaller in the Boreal Forest during the late period. In the Foothills, largest disturbed patch area (LDPA) converged on the HRV between the early and late periods in terms of area, but %LDP was about 44% lower than the HRV by the late period (U = 47, p < 0.01; D = 0.85, p < 0.01). The opposite trend was observed for the Boreal Forest; LDPA diverged from the HRV by area, but %LDP converged and was about 4% lower than the HRV by the late period (U = 2,139, p = 0.36; D = 0.2, p = 0.45).

The distribution of remnant levels contracted and shifted away from the HRV between the early and late periods (Figure 4.5). Mean island remnant area (MIRA) was smaller than the

Table 4.5 Median anthropogenic disturbance patterns by boreal region, period, and number of patches.

	Boreal Forest				Foothills			
	Early Period		Late Period		Early Period		Late Period	
Metric (x̃)	SPE	MPE	SPE	MPE	SPE	MPE	SPE	MPE
EA	1	55	1	97	2	80	1	7
MDPA	1	12	1	14	2	15	1	2
LDPA	1	27	1	40	2	31	1	4
%LDP	100%	46	100	41	99	45	100	54
MIRA	0	5	0	1	0	< 1	0	0
LIRA	0	7	0	1	0	1	0	0
%IR	0%	10	0	2	0	1	0	0
%TR	0%	42	0	30	1	33	0	16
ED	0.41	0.13	0.47	0.11	0.29	0.12	0.33	0.25

SPE = disturbance events comprised of one disturbed patch.
MPE = disturbance events comprised of more than one disturbed patch.

Figure 4.4 Anthropogenic disturbance event area (EA), largest disturbed patch area (LDPA), and mean disturbed patch area (MDPA) for multiple patch events (MPE) in the two study areas by period.

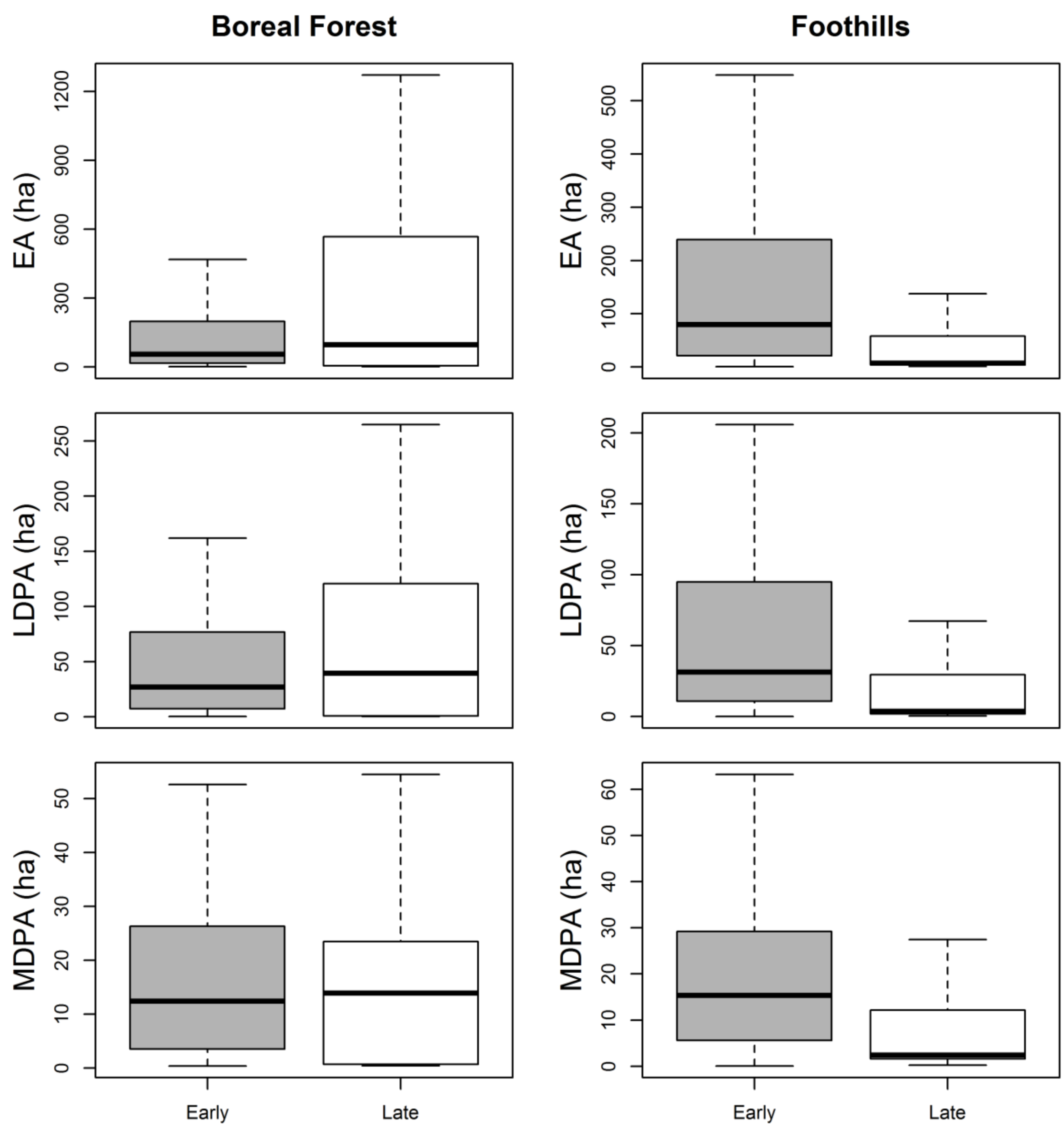
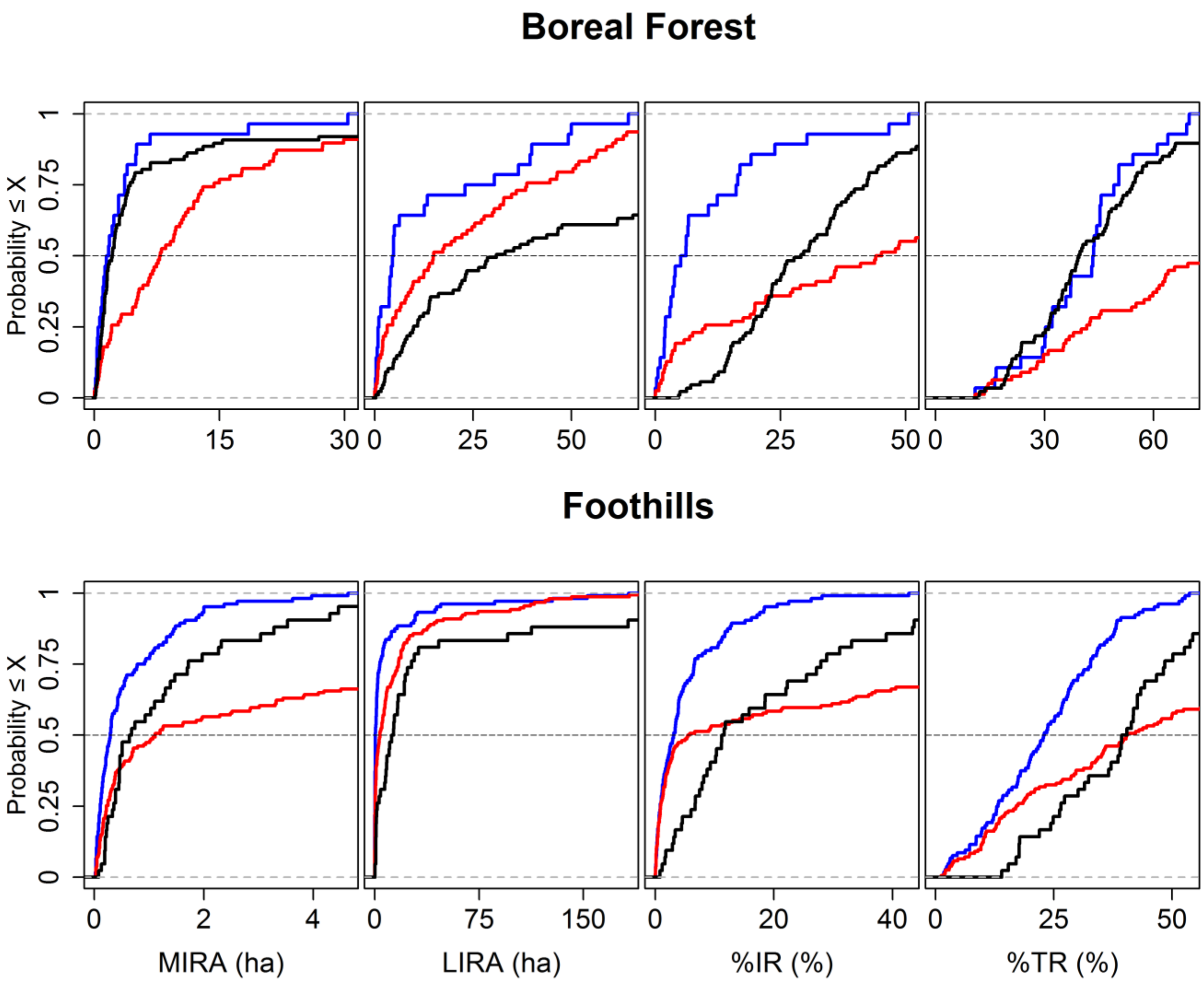


Figure 4.5 Cumulative distribution function of mean island remnant area (MIRA), largest island remnant area (LIRA), percent island remnants (%IR), and percent total remnants (%TR) for events with at least one island remnant in the Boreal Forest (top) and Foothills (bottom) during the early period (blue line) and the late period (red line) compared with the historical range-of-variability (black line).



HRV in the Foothills and Boreal Forest during the late period Forest. %IR was 8% lower than the HRV in the Foothills ($U = 854.5$, $p < 0.01$; $D = 0.53$, $p < 0.01$) and about 24% lower in the Boreal Forest by the late period ($U = 324$, $p < 0.01$; $D = 0.63$, $p < 0.01$). In the Foothills, %TR was about 13% lower than the HRV during the late period ($U = 938$, $p < 0.01$; $D = 0.5$, $p < 0.01$), but only 4% higher than the HRV in the Boreal Forest ($U = 1{,}178.5$, $p = 0.8$; $D = 0.14$, $p = 0.78$) (Figure 4.5).

4.3.4 Resource extraction period disturbance trends

Cumulative anthropogenic disturbance patterns varied considerably between the resource extraction periods that were examined. Generally, the range of disturbed patch size contracted and decreased from the early to the late period in the Foothills, which was concomitant with smaller disturbance events (Figure 4.4). By contrast, disturbed patch size remained at similar levels through both the early and late periods in the Boreal Forest (Figure 4.4). Edge density increased from 0.12 $km \cdot ha^{-1}$ during the early period to 0.33 $km \cdot ha^{-1}$ during the late period in the Foothills and 0.40 $km \cdot ha^{-1}$ to 0.46 $km \cdot ha^{-1}$ in the Boreal Forest. Foothills MDPA decreased by approximately 3 ha during the late period (Figure 4.4). In contrast, the range of LDPA and EA expanded in the Boreal Forest, while MDPA remained relatively unchanged during the late period (Figure 4.4). %LDP decreased by about 5% in the Boreal Forest during the late period ($U = 1{,}118$, $p = 0.02$; $D = 0.26$, $p = 0.06$). In the Foothills, %LDP increased by about 9%, from 91% to 100% between the early and late periods ($U = 21{,}920$, $p < 0.01$; $D = 0.22$, $p < 0.01$).

Remnant levels decreased between the early and late periods in terms of numbers, area, and percentage. In the Foothills, %IR decreased by about 3% ($U = 5{,}976$, $p < 0.01$; $D = 0.39$, $p < 0.01$) and %TR decreased by 18% ($U = 4{,}760$, $p < 0.01$; $D = 0.44$, $p < 0.01$). This trend was generally consistent across disturbances of all size classes, although %IR increased for disturbances between 1,000 and 10,000 ha (Figure 4.3). In the Boreal Forest, %IR decreased by 39% ($U = 451.5$, $p < 0.01$; $D = 0.56$, $p < 0.01$) and %TR decreased by 34% ($U = 538$, $p < 0.01$; $D = 0.54$, $p < 0.01$) (Figure 4.3).

4.4 Discussion

4.4.1 Oil and gas development drives anthropogenic patterns beyond the HRV

Overall, the results suggest that the anthropogenic disturbance patterns have been outside the HRV for several decades for both landscapes in the western Canadian boreal forest. Cumulative anthropogenic disturbance patterns differed in many ways from historical disturbance patterns: number of patches per event was greater; patch size was smaller; remnant levels were lower; and the number of single-patch events (SPE) was greater. Similarly, the percentage area of the largest disturbed patch within an event was consistently below the 70-90% average found in fires (Andison and McCleary, 2014). Furthermore, the gap between anthropogenic disturbance patterns and the HRV has increased over the last decade. This is particularly surprising considering recent efforts by forest managers in the study areas to implement HRV-based forestry practices, and suggests that the increase in energy sector activity may be overwhelming any concomitant change towards HRV from forest management.

The observed shifts in cumulative anthropogenic disturbance patterns are likely driven by oil and gas development. In 2002, forest disturbance related to the energy sector in the Boreal Forest was estimated at 11,000 ha · yr^{-1} compared with 16,000 ha · yr^{-1} for the forestry sector (Schneider *et al.*, 2003). Such levels of land use may have compromised forestry-based initiatives to emulate historical disturbance regimes (Pickell *et al.*, 2013). For example, although fires larger than 10,000 ha are responsible for most of the historical landscape patterns in the Boreal Forest, harvest events of this size are socially unacceptable and controversial (Carlson and Kurz, 2007). During the period that was studied, only one anthropogenic disturbance event in the Foothills attained a size class larger than 10,000 ha. Similarly, leaving significant levels of remnant vegetation on pine-dominated landscapes, such as those commonly found in the Foothills, contradicts provincially-led efforts to reduce the threat of mountain pine beetle (*Dendroctonus ponderosae*). The significant shift in disturbance patterns suggests that a shift in one sector of the economy can have dramatic outcomes on landscape structure – in this case the increase of the energy sector activity *circa* 2000. Other studies have reported similar results linking economic activity to forest disturbance and cover. For example, Masek *et al.* (2013) observed lower rates of forest disturbance in the U.S. during the recessions of 1990-1991 and 2000-2001, and higher rates of forest disturbance during 1997-2000, a period of economic growth measured by sustained increase in gross domestic product. Similarly, forest cover increased in Russia during the years following the collapse of the Soviet Union (Baumann *et al.*, 2012).

The transition of the anthropogenic disturbance regime that was detected is consistent with the current understanding of historical land management in the western Canadian boreal forest. In the Foothills, forestry activities were initiated in the 1950s, most of which created a significant legacy of first-pass "checkerboard" disturbance patterns that favored larger events, high levels of total remnants, and low levels of island remnants. Moreover, the transition from the early to the late period in the Foothills corresponds to a significant increase in the exploration and development of natural gas resources in that region. This land use shift favored single patch events (SPE) of high duration with no remnants. I found a 16-fold increase in small single patch events (from 35 per decade to 570 per decade) and an overall increase in SPE from 66% to 75% of all events between the early and late periods, The doubling of edge density for multiple patch events (MPE) in the Foothills is also consistent with the significant increase in energy sector activity associated with more seismic lines and roads. The impact of unconventional oil and gas extraction in the boreal forest is not well understood; however, the issue is positioned to be a critical area of research in the near future for Alberta and many more forested jurisdictions around the world (Northrup and Wittemyer, 2013).

4.4.2 Ecological considerations for multiple-use landscapes

Oil and gas development in Alberta was associated with declining remnant levels and increasing edge density, which both have significant implications for resilience of boreal forested landscapes. For example, more than 90% of the seismic lines in the study areas were cleared in the last 25 years, and more than 85% have been cleared since 2000. The widespread clearing of these seismic lines was likely a contributing factor to declining remnant size and levels during the late period. Undisturbed remnants represent ecological memory that supports the forest ecosystem in retaining its characteristic identity (Bengtsson *et al.*, 2003; Delong and Kessler, 2000; Peterson, 2002; Thompson *et al.*, 2001). The systematic eradication of ecological memory via high magnitude anthropogenic disturbance like roads, seismic lines, and well sites remains a concern for the ecological resilience of the region (Peterson, 2002). Similarly, well site construction severely modifies forest soils resulting in the loss of local seed and bud stores, and these oil and gas features may persist on the landscape for several decades before primary succession occurs and carbon sequestration re-initiates (MacFarlane, 1999; Osko and Macfarlane, 2001).

Most of the Canadian boreal forest is considered remote with less than 5% under some formal protection, but increased pressure from oil and gas extraction will likely "open up" more of the boreal forest to development in the future (Andrew *et al.*, 2014). Some authors have proposed intensive land management schemes for improving conditions on Canadian boreal forest landscapes (Binkley, 1997); however, the feasibility of their implementation remains an under-investigated topic (Tittler *et al.*, 2012). The greater economic activity generated by oil and gas extraction is likely to deflect efforts to develop effective conservation zonation, at least in Alberta. For example, the mean well site density in "protected areas" of Alberta was two times greater than the mean well site density for the province (Timoney and Lee, 2001).

Predators are often species at risk in human-dominated landscapes, and these species often play a critical role in regulating ecosystem production, function, structure, and composition (Woodroffe, 2000). For example, anthropogenic changes to landscape structure had adverse effects on populations of grizzly bear (*Ursus arctos*) in the Foothills (Linke *et al.*, 2005; Nielsen *et al.*, 2008) and wolf (*Canis lupus*) in the eastern (Lesmerises *et al.*, 2012) and central boreal forest (James and Stuart-Smith, 2000). The increased pressure placed on grizzly bear by intensive land uses makes the Foothills region especially vulnerable to a top-down trophic cascade. In contrast, anthropogenic disturbance can amplify some trophic relations, such as wolf predation of caribou (*Rangifer tarandus*) in the boreal forest (James and Stuart-Smith, 2000).

4.4.3 Considerations for designing anthropogenic disturbances

Considerable effort has been made over recent years to align the patterns of resource extraction activities with the HRV (Pickell *et al.*, 2013), but there remains room for improvement. For example, the largest 1% of fires routinely comprised greater than 97% of the area disturbed on the boreal forest landscape (Cumming, 2001a), but the largest 1% of anthropogenic disturbances in the same region only comprised only 51% of the disturbed area. Emulating the upper size limit of the HRV will be challenging to implement for social reasons and still more research is needed to provide the ecological justifications at finer scales.

The duration or persistence of some anthropogenic disturbance events can be reduced through strategic restoration activities. The NEPTUNE model has the potential to forecast oil and gas recovery scenarios given different assumptions about reclamation practices. For example, I assumed a fixed 40 year legacy for seismic lines and well sites based on current research and practices, but these parameters could be varied to optimize specific cumulative pattern metrics for HRV targets. Such research could also test hypotheses around erasing the footprint of the energy sector in Alberta using HRV-based forest harvesting or reintroduction of fire into the landscapes. However, given known recovery times for oil and gas disturbances as well as current production by active well sites, it is unlikely that forest harvesting could erase the energy sector footprint during a single rotation in the near future. Moreover, there are many challenges associated with reintroducing fire into these industrialized landscapes (Hirsch *et al.*, 2001).

The research demonstrates the limited degree to which forestry alone can affect change on the landscapes. The analyses indicate that several notable metrics are prone to amplification due to cumulative anthropogenic disturbance between industrial stakeholders. For example, increasing the amount of linear corridors (*i.e.*, roads, rails, pipelines, cut lines and transmission lines) in a disturbance can increase the number of island remnants and decrease mean island remnant size. In the Foothills, edge density increased by approximately 58% between the early and the late period and this trend was associated with a 74% reduction in mean island remnant size at the median. Anthropogenic disturbances should be designed to reduce the amount of edge habitat created by linear corridors, which increase the propensity for disease and invasive species to spread. Only in the recent decade has cumulative anthropogenic disturbance mapping been realized in Alberta (*e.g.*, Gaulton *et al.*, 2011; Linke and McDermid, 2012; Pasher *et al.*, 2013; Pickell *et al.*, 2013) with previous research developed around models and projections (*e.g.*, (Schneider *et al.*, 2003; Yamasaki *et al.*, 2008). Beyond on-going HRV initiatives, synergistic planning and cooperation among the resource extraction sectors in Alberta will be necessary to reduce the gap between anthropogenic disturbance patterns and the HRV.

Chapter 5

Monitoring anthropogenic disturbance trends in an industrialized boreal forest with Landsat time series

5.1 Introduction

Humans now appropriate a greater fraction of the terrestrial surface of Earth than ever before (Imhoff *et al.*, 2004), in particular of forested ecosystems, which contain the highest density of biomass of all ecosystems and provide a wide variety of ecosystem goods and services to humanity (Millennium Ecosystem Assessment, 2005). Mapping anthropogenic changes to forest cover is essential to monitoring landscape condition. Forested landscapes in North America have undergone significant changes in forest cover due to the extraction of energy and mineral resources (Pickell *et al.*, 2014a) as well as extraction of timber resources and long-term conversion to other land uses. Such activities have altered ecosystem structure and function (Simmons *et al.*, 2008) and significantly reduced biodiversity (Butt *et al.*, 2013).

The extent of human modification of forest cover in North America remains relatively unknown. Pasher *et al.* (2013) estimate the footprint of anthropogenic disturbance in the Canadian boreal zone to be approximately 24 million hectares, most of which was attributed to forest harvesting. Their mapping efforts were undertaken using manual interpretation, which is both costly and time-consuming, especially for large areas such as the Boreal. Efficiently monitoring human modification of the environment requires robust automated methods and remote sensing is well-positioned to meet these mapping needs (Powers *et al.*, 2015).

The Landsat satellite program has continually overflown the planet every 16 days for the last 42 years. The open release of the Landsat image archive to the public in 2008 (Woodcock *et al.*, 2008) has spawned numerous advancements in image processing and automated change detection methods (Wulder *et al.*, 2008c). As a result, every available Landsat image is potentially able to be integrated into forest cover change assessment at local to global scales (Hansen *et al.*, 2013).

Spectral trend analysis approaches can be used to track changes in surface reflectance from time series of Landsat imagery. This approach takes advantage of three properties of energy exchange and forest dynamics when detecting changes through time: (1) disturbed vegetation is spectrally dissimilar from healthy vegetation, particularly in the mid- and near-infrared bands; (2) disturbed vegetation takes several years to recover; and (3) persistence of a trend through time can be used to attribute a class of disturbed, forest, or non-forest. Spectral trend analysis has been implemented in several automated algorithms such as LandTrendr (Kennedy *et al.*, 2010) and the highly autonomous vegetation change tracker (VCT) (Huang *et al.*, 2010). The VCT was recently used to estimate recent forest disturbance trends in the United States and shows promise for detecting multiple types of disturbance (Masek *et al.*, 2013).

In this chapter, I present an examination of a spectral trend analysis approach in a western Canadian boreal forest that has undergone significant resource development and human modification. The western Canadian boreal forest is an ideal location to apply automated detection of anthropogenic disturbance due to the high severity and extent of resource development in the region. I applied the VCT algorithm to detect anthropogenic disturbances with the objectives of (1) discriminating anthropogenic from fire disturbances over a 28-year period and (2) quantifying the contribution of anthropogenic disturbance to landscape dynamics where oil and gas development has increased significantly over the last two decades.

5.2 Materials and methods

5.2.1 Study area

The study area was a single Landsat WRS-2 path-row (path45/row23) located in the Rocky Mountain foothills of Alberta, Canada. The terrain is gently undulating with elevation ranging from 500 m.a.s.l. to 1500 m.a.s.l. Approximately two-thirds of the study area is forested (Wulder *et al.*, 2008a), which is primarily evergreen forest comprised of *Picea*, and *Pinus* species. The forests in the area have been actively managed and harvested since 1955. Approximately half of the study area is reserved for protected areas while the remaining forest lands are actively managed. Large deposits of coal and natural gas underlay most of the forest cover in the region and recent advancements in recovery technologies have allowed for unprecedented rates of extraction since *circa* 2000.

5.2.2 Satellite time series data

Landsat surface reflectance (SR) data were acquired annually during the growing season (152 < day-of-year < 273) between 1984 and 2011. Landsat images were preferentially selected to minimize cloud cover with acquisition dates later in the growing season. Landsat TM acquisitions were preferred over Enhanced Thematic Mapper Plus (ETM+), and ETM+ Scan Line Corrector off acquisitions were not included in the time series analysis. Forest inventory data were acquired for a forest management zone, which included anthropogenic forest changes such as roads, pipelines, well sites, and forest harvesting since 1955. The inventory data were derived from standard interpretation of aerial photographs that are collected on an on-going basis for forest management objectives (Alberta ESRD, 2005). In addition, I used the Alberta Historical Fire Database (Alberta ESRD, n.d.) to identify the perimeters of fires that occurred in the study area during the study period. The fire polygons were collected from interpretation of post-fire aerial photographs.

All data were filtered to a common minimum mapping unit (MMU) to make the data compatible and minimize noise while still being able to detect small disturbances in the final disturbance map product. The forest inventory data were mapped to Alberta Vegetation Inventory standards (Alberta ESRD, 2005) at a MMU of 1 ha for anthropogenic features; the fires were mapped at 0.01 ha MMU, but only fires larger than 1 ha were used; and changes from the Landsat time series were mapped to a 12-pixel (~1 ha) MMU. The Landsat images were used to detect changes in the time series while the forest inventory data and fire perimeters were used to train the classification and assess the quality of the change detection procedure.

5.2.3 Overview of methods

A combination of two methods were undertaken to quantify the contribution of anthropogenic land cover change in the study area. First, forest disturbance was detected from a Landsat time series. The outcome of this procedure was an estimated year of disturbance for pixels classified as disturbance. Second, disturbed pixels were filtered by a MMU, converted to objects, and classified as either resource extraction or fire using a suite of descriptive attributes. Both processes are described in more detail below.

5.2.4 Change detection procedure

The VCT (Huang *et al.*, 2010) was applied for mapping forest disturbances using the disturbance index (DI) (Healey *et al.*, 2005). The DI utilizes a linear combination of the Tasseled Cap Transformation (TCT) that normalizes each pixel value to a dense forest class (Masek *et al.*, 2008). Significant and consistent deviations from the dense forest class are then classified as disturbance and the year of disturbance is recorded. Pixels that remain spectrally similar to the dense forest class throughout the time series are classified as persisting forest and pixels that are spectrally dissimilar are persisting non-forest.

A dense forest training mask was created annually for each Landsat scene by visually inspecting dense vegetation cover during the year 2000 from the version 5 MODIS vegetation continuous field (VCF) product collected in 2000 (DiMiceli *et al.*, 2011), normalized differenced vegetation index (NDVI) values, and a true color composite. The NDVI and VCF values for dense forest identified in the true color image in 2000 were then used as thresholds for creating the dense forest mask for each image in the time series.

Once a forest mask was created, the DI was calculated at annual time steps for each Landsat image. In order for a pixel to be flagged as disturbed, the DI trend had to exceed an upper and lower threshold for a designated number of observations. A sensitivity analysis was performed from initial threshold values based on previous research with the VCT in western U.S. forests (Masek *et al.*, 2013). Final threshold values were selected to reduce single pixels detected as disturbance and overall noise while preserving the spatial and temporal integrity of the mapped disturbance events. Pixels could be disturbed more than once during the time series, but only the most recent year of disturbance was recorded. Pixels that contained cloudy observations for more than half of the time series or more than five consecutive years were not classified. Cloudy values were interpolated linearly when good observations were available for previous and subsequent years. Finally, a water mask based on near-infrared reflectance was applied to the time series analysis to classify persistent water bodies.

5.2.5 Disturbance type classification

In the second step, I differentiated between resource extraction and fire disturbances. A subset of the forest inventory data and fire maps were randomly divided into two equal-number samples to be used to train and evaluate the classification. Approximately 50% of the objects were used to train the classification and 50% were used to evaluate the classification. The fire sample (19,022 ha) was randomly drawn from the entire Landsat path-row, while the resource extraction sample (94,588 ha) was randomly drawn only from the forest management zone.

Disturbances were classified using a set of object-based descriptive attributes derived using FETEX 2.0 software (Ruiz *et al.*, 2011) describing the geometrical, spectral and textural properties of the disturbances. The geometry and shape was described using area, perimeter, compactness (Bogaert *et al.*, 2000), shape index (McGarigal and Marks, 1994), and fractal dimension (Krummel *et al.*, 1987). Spectral attributes were computed from the spectral bands and from common indices: DI; normalized burn ratio (NBR) (Key and Benson, 2006); NDVI (Tucker, 1979); and greenness, wetness, and brightness from the TCT (Crist, 1985). The mean and standard deviation of the indices and the differences of spectral means between the pre- and post-disturbance years were computed. The texture descriptors extracted were: edge density (Sutton and Hall, 1972); gray level (Haralick *et al.*, 1973); skewness and kurtosis; and experimental semivariogram indices (Balaguer-Beser *et al.*, 2013). These attributes were computed from the near-infrared band (Landsat band 4) during the disturbance year.

Classification was performed using linear discriminant analysis (LDA). LDA is a multivariate statistical technique where the dependent variable is categorical, whilst the independent variables are continuous and are used to determine the class to which the objects belong (Everitt and Dunn, 2001; Huberty, 1994). This technique maximizes the variability between groups based on the continuous variables, while minimizing variability within groups.

5.2.6 Disturbance classification accuracy assessment

The spatial accuracy of the disturbance classification was assessed using an error matrix (Congalton, 1991). In addition to the overall accuracy, user's and producer's accuracies were calculated per class, which respectively measure the commission and omission errors. First, the reference data (mapped anthropogenic features and fires) were used to assess the area-based accuracy of the disturbed and non-disturbed classes generated from the time series analysis. Next, the temporal accuracy was assessed by inspecting a scatterplot of the reference data year and the VCT classified year. A final assessment was undertaken to scrutinize the accuracy of the disturbance type classification (*i.e.*, resource extraction vs. fire) using the reference fire database and forest inventory spatial dataset. The sample of objects that was used for training was excluded from the full set of objects that were used in the evaluation.

5.3 Results

The temporal quality of the disturbance classification was high, with the majority of errors within one year relative to the forest inventory date for approximately 89% of disturbed forest by area (Figure 5.1). Most errors in the temporal domain resulted from the VCT classifying disturbances one year after the known disturbance date. An error matrix of the disturbance classification is presented in Table 5.1 and disturbance type classification is presented in Table 5.2. Overall accuracy for the area-based disturbance classification was about 93% and overall accuracy for the object-based disturbance type classification was approximately 94%. Errors of omission and commission were highest for the fire class (55% and 63% accuracy, respectively) which was also the class with the fewest samples (Table 5.2). User's accuracy for the disturbed class was about 76%. The net committed errors (rate of false positives) and omitted errors (rate of false negatives) of the disturbed class were approximately 15% and 24%, respectively. The net committed and omitted errors for the fire class were 44% and 37%, respectively. The most important descriptive attributes for classifying disturbance type were mean NBR, change in NBR, band 5 (mid-infrared spectral band), band 2 (visible green spectral band), DI, and greenness.

Figure 5.1 Scatterplot of disturbance year attribution by the vegetation change tracker (VCT) compared with forest inventory. Values above the *y* = *x* line indicate that the VCT disturbance year is later than the inventory disturbance year.

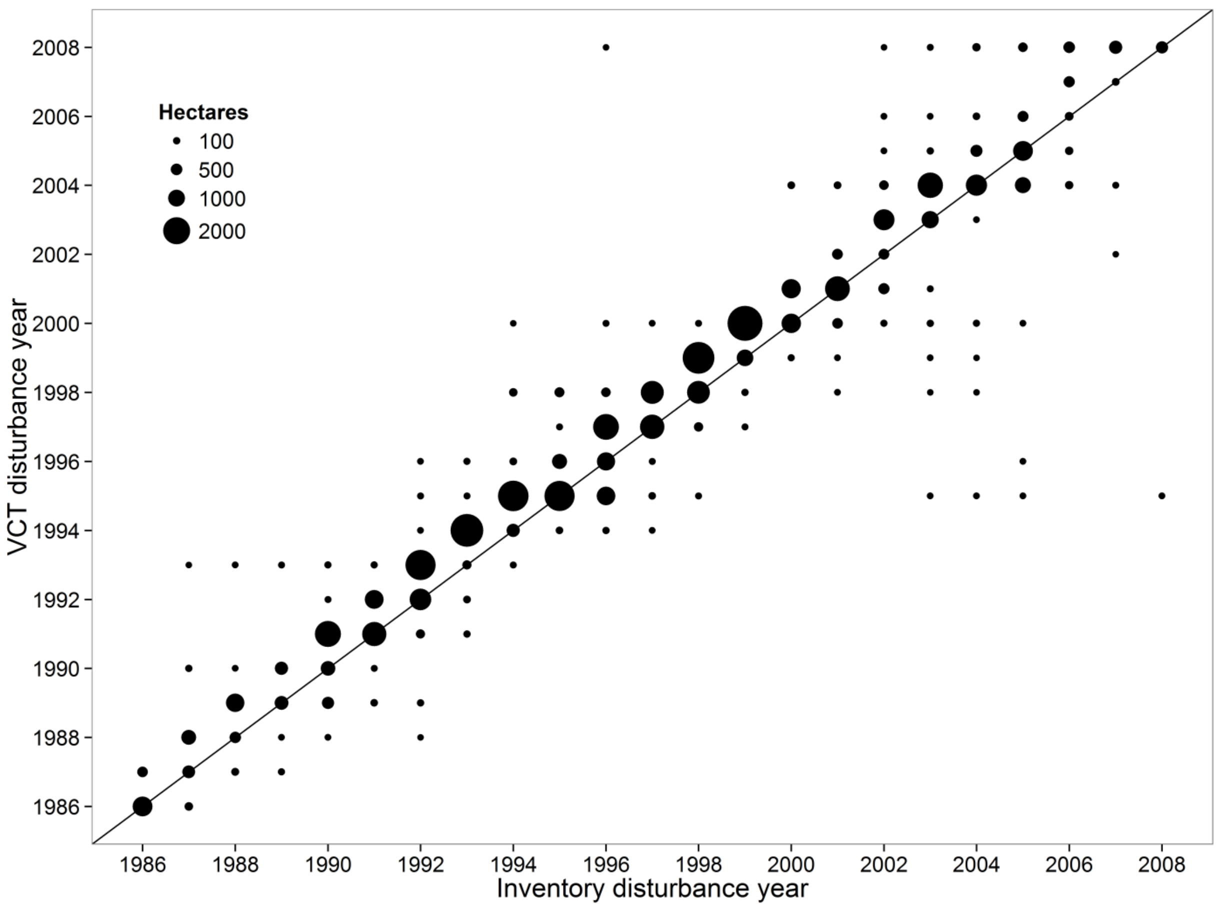


Table 5.1 Error matrix of disturbance classification.

	Reference data (ha)		
VCT map (ha)	**Disturbed**	**Non-disturbed**	**User's accuracy (%)**
Disturbed	86,289	27,565	75.8
Non-disturbed	15,357	474,267	96.8
Producer's accuracy (%)	84.9	94.5	92.9 (overall)

Table 5.2 Error matrix of disturbance type classification.

	Reference data (n objects)		
VCT map (n objects)	**Fire**	**Resource extraction**	**User's accuracy (%)**
Fire	228	135	62.8
Resource extraction	183	4592	96.2
Producer's accuracy (%)	55.5	97.1	93.8 (overall)

Disturbance rates varied temporally and spatially across the region. The rate of disturbance increased between 1986 and 2000, and declined since. Disturbance was mostly clustered in the north within the primary forest management zone (Figure 5.2). Taken across the entire time series, the average annual rate of disturbance was approximately 6,555 ha · yr^{-1} (σ = 6,148 ha). The large standard deviation was primarily a result of two outlying years (2003 and 2004) with much higher levels of disturbance (Figure 5.3). With those years removed, the average annual rate of disturbance declined to approximately 4,780 ha · yr^{-1} (σ = 1,865 ha), which is likely more representative of the landscape and region in general. The sharp increase in disturbance during 2003 was primarily attributed to the Syncline Ridge fire that burned nearly 28,000 ha in Jasper National Park. The sharp increase in resource extraction disturbance the following year (2004) was likely a result of misclassification of post-fire mortality of the Syncline Ridge fire and other fires as resource extraction disturbance and poor disturbance year attribution by the VCT due to extensive cloud cover in those years.

Resource extraction disturbance rates were relatively stable across the time series and accounted for the majority of disturbance in any given year (Figure 5.3). The average annual rate of resource extraction disturbance was approximately 5,640 ha · yr^{-1} (σ = 4,362 ha) across the entire time series and 4,630 ha · yr^{-1} (σ = 1,844 ha) with 2003 and 2004 excluded. Approximately 91% of all disturbances were resource extraction and approximately 86% of total disturbed area was resource extraction. The region underwent significant development of oil and gas resources beginning in the early 2000s. The year 2004 showed the greatest levels of anthropogenic disturbance and a large portion of the landscape was impacted by road and well site construction (Figure 5.2). However, the majority of resource extraction disturbance during that year was driven primarily by forest harvesting and subsequent annual resource extraction disturbance rates did not increase.

5.4 Discussion

While the method demonstrated good success with detecting and classifying disturbances larger than one ha, there are many finer-scale disturbances in the study area that do not contribute

Figure 5.2 Forest harvesting, roads, and well sites in the north of Landsat tile path45/row23 within an intensive forest management zone.

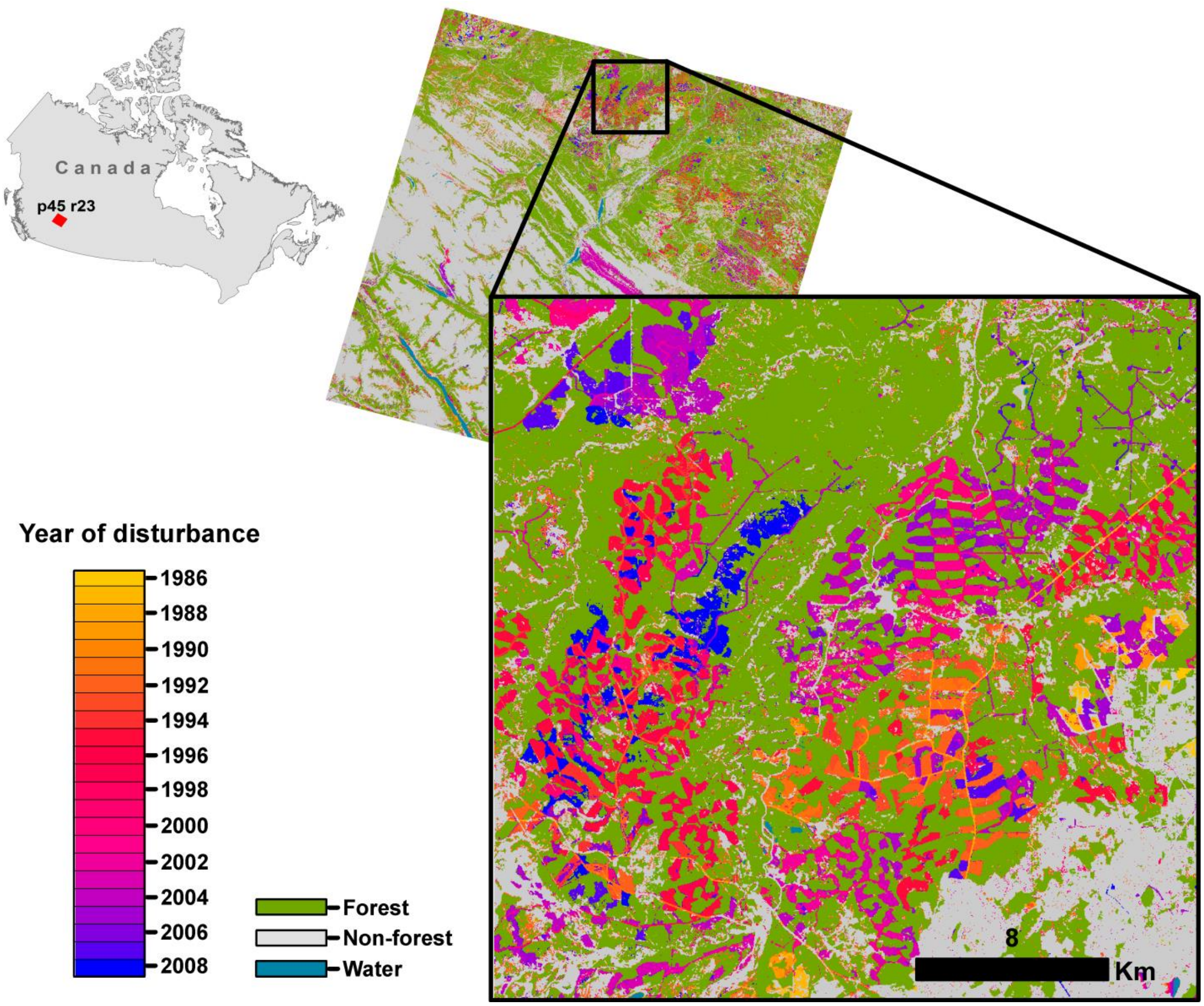
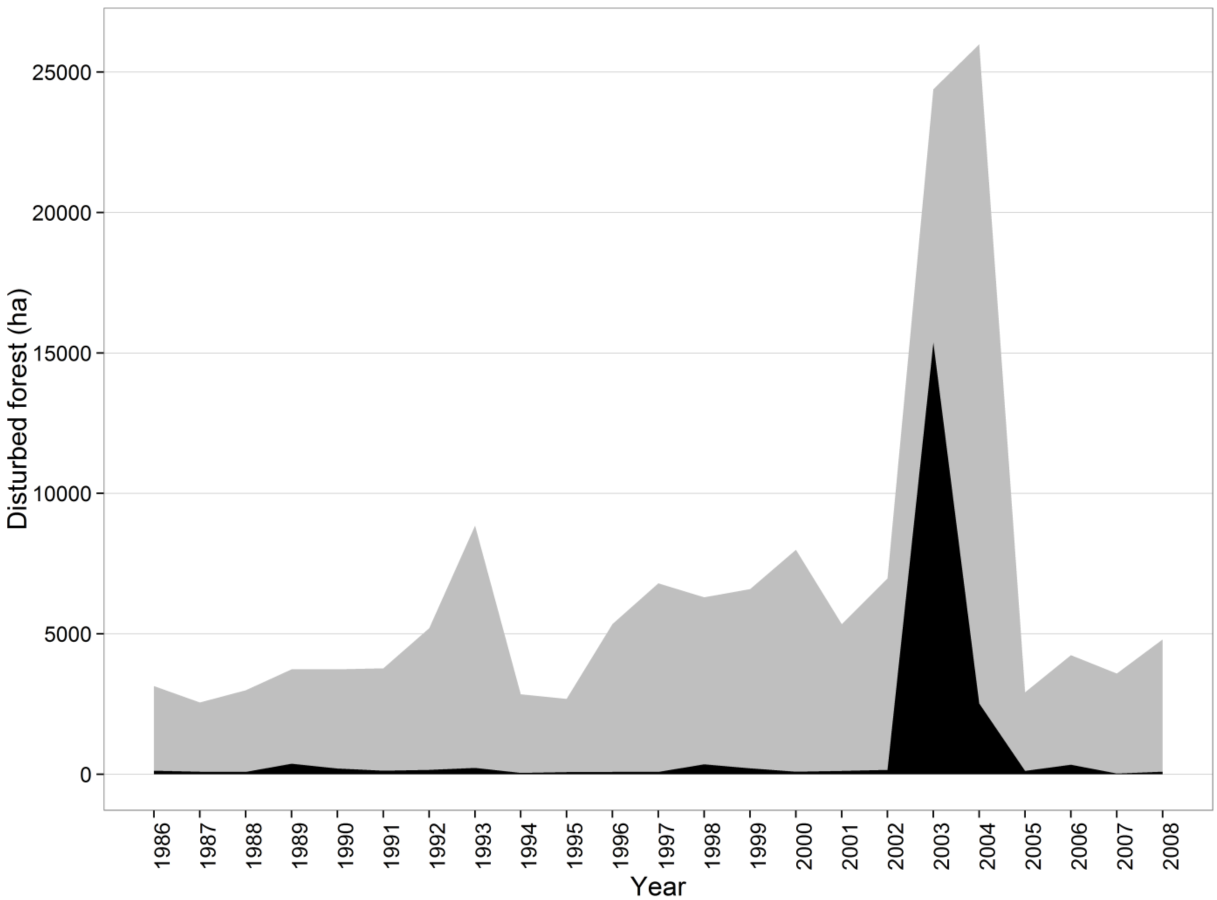


Figure 5.3 Forest disturbance trends for the study area: resource extraction disturbance (gray) and fire (black).



greatly to disturbance area, but have large impacts on the boreal forest. For example, the density of seismic lines in the region is among the highest in the province of Alberta, but these features are below the spatial resolution of the Landsat imagery that was used (Powers *et al.*, 2015). A classification of other disturbance types like roads and seismic lines would require finer spatial resolution imagery, but there is an inherent trade-off between detecting smaller features (spatial resolution) and sampling large areas (extent) for disturbance (*i.e.*, spatial scale).

The modest discrepancies in the disturbance date attribution may have been related to errors in the inventory data and data gaps in the imagery in some years. Harvesting occurs on an on-going basis in the study area and it may take multiple years before a harvest feature is considered complete in the forest inventory data. The harvest completion date was used to assess the temporal accuracy and thus some disturbances could have occurred before this date. Additionally, cloud, haze or shadow could cause the disturbance to be attributed the following cloud-free year in the time series.

Overall, the disturbance classification accuracy was within the range of previous classification accuracies reported for the VCT (Thomas *et al.*, 2011). The results suggest that the VCT was more likely to underestimate than overestimate the area of disturbance. The relatively low producer's accuracy of the fire class suggests that there are still some challenges associated with discriminating disturbance types which may be related to the structural characteristics of fire and typical anthropogenic disturbance. For example, shadowing and soil exposure vary between the classes, and also within fires, and the patchy nature of fires potentially affected the ability of the VCT to identify the heterogeneous canopy mortality of fires.

The results show that resource extraction disturbed more forested area than fire for the study area. Fire is believed to be the primary driver of landscape structure and forest change in the boreal zone as a whole (Weber and Flannigan, 1997), however the study area, in the south-west of the boreal, was settled in the 1950s and fire suppression has been effective at reducing area burned in the region during the study period (Cumming, 2005). The objective of contemporary forest management in the study area has been to replace the fire disturbance regime with forest harvesting while keeping the combined rates of disturbance similar to pre-settlement levels (Bergeron *et al.*, 2002). Other non-anthropogenic disturbances such as wind throw and insect attack were not included in the classification due to the lack of available spatial reference data for these disturbances in the study area. However, given the levels of disturbance observed for the study area, these disturbance types were not likely to significantly alter the contribution of anthropogenic land cover change. In addition, the VCT algorithm is not designed to detect slow degradation or subtle disturbance events that only affect a small fraction of the forest canopy.

Chapter 6

Managed boreal forests drive rapidly changing landscape patterns over the last three decades in Canada

6.1 Introduction

The relationship between ecological processes and spatiotemporal heterogeneity has long been a critical topic of ecological research (Gustafson, 1998; Haines-Young and Chopping, 1996). Landscape pattern indices have been developed to characterize valuable ecological patterns (Gergel, 2007). For example, fragmentation of forest cover captures habitat loss, habitat perforation, and connectivity (Fahrig, 2003). As a consequence, the spatial and temporal context of landscape patterns has emerged as a research priority in conservation science (Cardille *et al.*, 2005).

Patterns associated with landscapes present challenges for applying rigorous statistical analyses: spatial data of sufficient quality is not always available or consistent across jurisdictions, samples tend to be autocorrelated (*i.e.*, not independent), and landscapes are unique natural experiments that are challenging to replicate. Several solutions have been proposed to overcome these limitations including the development of comprehensive databases of landscape patterns from satellite imagery (Cardille *et al.*, 2005). Large area land cover maps have already been utilized for fragmentation analyses in Canada (Wulder *et al.*, 2008d) and the United States (Cardille *et al.*, 2005); however, these analyses have been limited to a single time period.

Advancements in automated remote sensing techniques have provided the means to map large areas from regional to global scales (Hansen *et al.*, 2013; Powers *et al.*, 2015). Large area mapping of forest cover changes has resulted in new opportunities to investigate the relationship between disturbance processes and spatiotemporal heterogeneity (Fry *et al.*, 2011; Hansen *et al.*, 2013; Homer *et al.*, 2007; Jin *et al.*, 2013; Vogelmann *et al.*, 2001; Wulder and Nelson, 2003). Forest cover maps provide a unique opportunity to understand how patterns of landscape structure vary spatially and the potential consequences of changing landscape structure for habitat availability and loss. For example, the METALAND project represents the largest database of landscape pattern indices derived from the *circa* 1992 National Land Cover Database for the conterminous United States (Cardille *et al.*, 2005). Similarly, the work by Wulder and colleagues (2011, 2008d) utilized the Earth Observation for Sustainable Development of Forests land cover map of Canada to summarize fragmentation of boreal forests *circa* 2000. Such research enhances the ability to identify representative landscapes across broad scales and to characterize the spatial and statistical context of landscape pattern indices (Cardille *et al.*, 2005, 2012). The behavior of landscape patterns remains to be characterized for large areas through time and at multiple scales.

Some of the questions initially put forward by Cardille *et al.* (2005) stemmed from limited access to comprehensive historical data at the time. Since 2008, the entire Landsat archive has been opened with free access (Woodcock *et al.*, 2008), which has significantly advanced progress in time series analysis over the current decade. Access to new data and technology allow us to ask and answer very basic, but pressing questions in spatial ecology: How have spatial patterns changed over time? What are the drivers of spatial patterns? What is the historical context for assessing present-day spatial patterns?

Boreal forests represent good opportunities for testing hypotheses about temporal changes in landscape patterns because they are easily distinguished from other non-forest land cover classes and provide a wide range of ecosystem goods and services such as habitat, carbon sequestration, and wood fibre (Shvidenko *et al.*, 2005). Moreover, boreal forests are particularly sensitive to climate change and state transitions because most boreal tree genera are limited by temperature and photoperiod (Chapin *et al.*, 2010; Scheffer *et al.*, 2012). Large even-aged patches of forest are emergent keystone landscape structures in boreal forests as a consequence of the inverse relationship between fire size and frequency (Bergeron *et al.*, 2007; Cumming, 2001a). Yet there is also significant structural and compositional diversity at finer scales given the high prevalence of residual vegetation that are readily observed by remotely sensed imagery (Andison, 2012; Eberhart and Woodard, 1987; Pickell *et al.*, 2013). Lastly, intensive resource extraction has in some areas created a novel overlay of structures in addition to various sources of inherent variation such as terrain (Pasher *et al.*, 2013; Pickell *et al.*, 2013).

The degree to which boreal landscape patterns have changed in recent years is largely unknown, but such research is of interest to resource managers in the Canadian boreal forest where an emerging objective of resource extraction planning is to emulate historical landscape conditions (Hunter, 1993; Landres *et al.*, 1999). This approach to resource development is the cornerstone of ecosystem-based management (EBM) and has been implemented in several areas in the boreal forest across Canada (Pickell *et al.*, 2013). The rationale is that historical landscape conditions maintained higher levels of biodiversity and habitat for a wide range of species (Drever *et al.*, 2006; Hunter, 1993). Delimiting the baseline condition, known as the historical range-of-variability, is a requisite for EBM research (Landres *et al.*, 1999; Morgan *et al.*, 1994). The historical range-of-variability can be used as a benchmark to assess current landscape condition by providing a population of landscapes that is suitable for robust statistical analysis (Cardille *et al.*, 2005). Such a benchmark is also necessary as the baseline for creating models for detecting deviations from the historical range-of-variability due to climate change, as well as predicting future landscape conditions under different climate scenarios (Flannigan *et al.*, 1998).

In this chapter, a methodology is introduced for the enumeration of a landscape pattern database for the boreal forest of Canada derived from time series Landsat satellite imagery. The primary objective was to quantify the recent high-magnitude changes to the forest landscape patterns and populate a database of landscape pattern indices that could be used to assess the historical trends of boreal forest landscape patterns in Canada. Finally, I demonstrate how the database and indicators can be used together to identify regions of the boreal forest undergoing significant changes to forest cover abundance and configuration.

6.2 Materials and methods

6.2.1 Sampling design and data acquisition

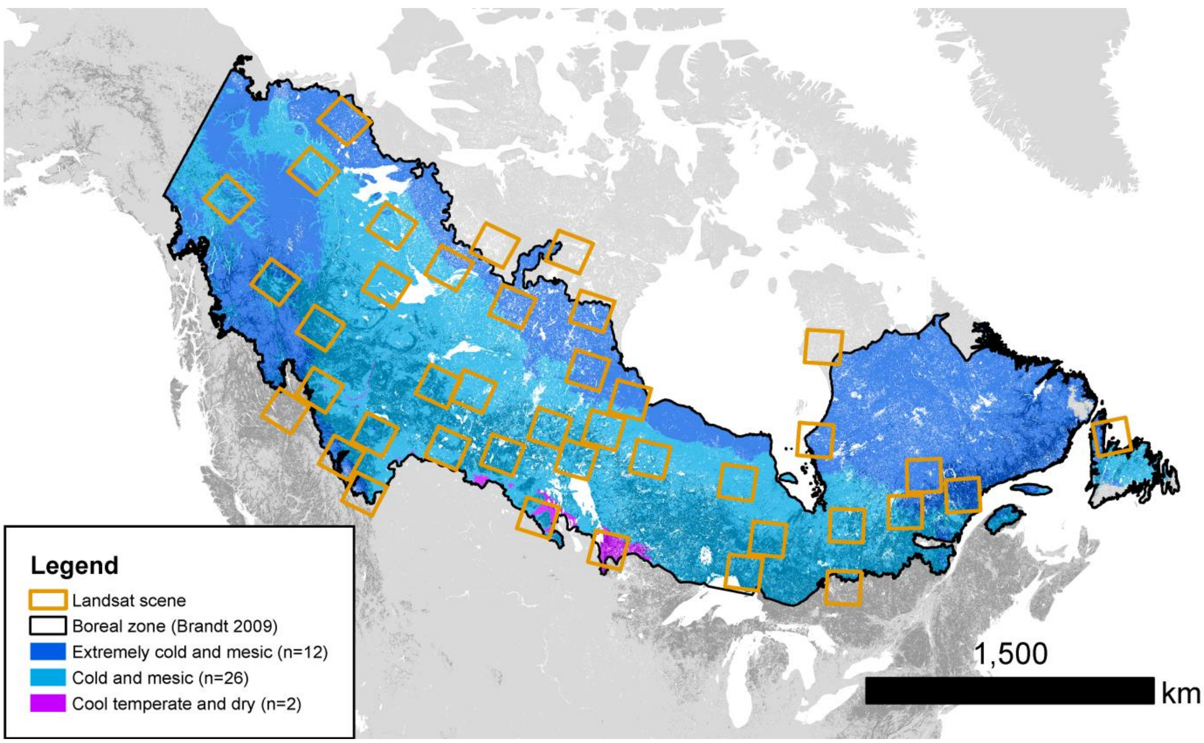
A sampling frame was defined for the Canadian boreal zone (Brandt, 2009) based on intersecting WRS-2 Landsat path-rows (hereafter referred to as scenes) and forest cover (Sexton *et al.*, 2013). Scenes were ranked based on the global continuous tree cover fields developed by Sexton and colleagues (2013) and scenes above the 5th percentile of forest cover were included in the population sampling frame as a final sample of n = 40 scenes stratified by major bio-climatic zones (Metzger *et al.*, 2013) within the Canadian boreal zone (Brandt, 2009) (Figure 6.1). All Landsat images were acquired during the growing season and neighbor overlapping scenes with less than 70% cloud cover were downloaded from the archive for each targeted scene. The growing season was defined uniquely for each Landsat scene using two attributes: length-of-season was derived from the scene center latitude of each Landsat scene (Zhou *et al.*, 2001); and the start- and end-of-season days were defined equally distant from July 28, which was considered the peak normalized differenced vegetation index (NDVI) day-of-year (Zeng *et al.*, 2011). Defined in this manner, the start-of-season varied from May 7 to June 15 and the end-of-season ranged from September 9 to October 18 depending on the latitude of the Landsat scene.

Each pixel from all images in a given year was scored based on best available pixel (BAP) criteria: imaging sensor (Thematic Mapper or Enhanced Thematic Mapper Plus); cloud shadow and proximity to cloud and cloud shadow as a Gaussian function of a 50 m buffer; day-of-year as a Gaussian function of the growing season; and atmospheric opacity (White *et al.*, 2014). Pixels with the highest scores were then composited into annual BAP images for the change detection procedure described in the next section. For some years and locations, there was no best available pixel due to data gaps on all of the available images. In these cases, the spectral information was interpolated between the most two recent years with data. Pixel time series with three consecutive years of no data or with more than six total years of no data were excluded from further analysis.

6.2.2 Mapping forest cover changes

The process for mapping forest cover changes involved training an annual forest mask and then applying spectral trend logic to the trajectory of each pixel to derive the pixel state. There were three possible states for each pixel: persisting forest, persisting non-forest, and forest cover change. Persisting forest and persisting non-forest indicated that the pixel was forested or non-forested for all annual images. Forest cover changes were characterized by the change of a pixel from a forested state to a non-forested state and vice-versa. The forest cover changes were mapped during the study period (1985 to 2010) using a modified version of the Vegetation

Figure 6.1 Distribution of the Landsat scenes stratified across major bio-climate zones (Metzger *et al.*, 2013) within the North American boreal forest (Brandt, 2009). The bio-climate zones are transparent relative to the 2000 Moderate Resolution Imaging Spectroradiometer (MODIS) Vegetation Continuous Fields (VCF) layer depicting pixels greater than 50% forested (dark gray).



Change Tracker (VCT) (Huang *et al.*, 2010) for each target Landsat scene (Pickell *et al.*, 2014b). Previous studies have evaluated the accuracy of the VCT across a wide range of forest types including boreal forests and concluded that the VCT generally underestimates disturbance (*i.e.*, higher omission error rate) with the majority of disturbance events correctly identified to within ±1 year (Pickell *et al.*, 2014b; Thomas *et al.*, 2011). Overall accuracies for the VCT using change and no change categories have been reported to exceed 90% (Pickell *et al.*, 2014b; Thomas *et al.*, 2011).

Forest cover masks were trained using the Moderate resolution Imaging Spectroradiometer (MODIS) Vegetation Continuous Fields (MOD44B) product from the year 2000 (Sexton *et al.*, 2013), annual NDVI values, and visual interpretation of annual true-color images. The disturbance index (DI) (Healey *et al.*, 2005) was used for mapping forest cover changes with thresholds that were developed specifically for boreal forests (Pickell *et al.*, 2014b). The DI is a linear combination of the brightness, wetness, and greenness components of the Tasselled-Cap Transformation (Crist, 1985; Healey *et al.*, 2005). Pragmatically, DI values that deviate significantly from zero (*i.e.,* positively or negatively) are more likely to be non-forested (Healey *et al.*, 2005). Pixels that exceeded the DI threshold for at least three consecutive years with a DI change magnitude greater than five were labeled as *changed*. I only considered stand-replacing disturbance from fire and forest harvesting because forest structure changes significantly following greater magnitude disturbance severities and vegetation recovery can therefore be robustly characterized. Pixels that consistently exceeded the DI thresholds for the length of the time series were labeled as *persisting non-forest* and pixels that were within the DI thresholds were labeled *persisting forest*. The final forest cover maps were filtered to a minimum mapping unit (MMU) of one ha using the eight-neighbor pixel rule to ensure that the effect of map misclassification errors on landscape pattern indices was minimized (Langford *et al.*, 2006). Non-forest patches smaller than the MMU were reclassified as forest and forest patches smaller than the MMU were reclassified as non-forest. Pixels disturbed in multiple years were relatively infrequent during the 26-year time series and generally below the target MMU of one ha, so for simplicity only the most recent disturbances were considered in the analysis.

In order to account for forest cover gains over time, forest recovery was defined spectrally to be consistent with the forest disturbances. A suite of spectral indices were selected to represent a range of spectral responses of vegetation recovery: NDVI; Normalized Burn Ratio (NBR) (Key and Benson, 2006); Tasseled Cap Greenness (TCG) (Crist, 1985); and the mid-infrared spectral band of Landsat (band 5) (Schroeder *et al.*, 2011). Recovery was defined as 80% of the mean spectral value of the two years prior to disturbance for each disturbed pixel in the time series. This definition is similar to other spectrally-based forest recovery definitions (*e.g.*, Baumann *et al.*, 2012; Kennedy *et al.*, 2012; Schroeder *et al.*, 2011). Two years of pre-disturbance observations represented a compromise between characterizing recovery trends for as much of the time series as possible while using stable pre-disturbance trends that were reliable and representative of characteristics present at the pixel level. Thus, recovery was not quantified for disturbances occurring in the last year of the time series. The recovery trends were computed using the annual BAP composite images containing the original spectral information. The dynamic recovery model ensured that landscape pattern indices would not be biased in later years due to cumulative forest cover losses (*e.g.*, over-estimating landscape fragmentation).

6.2.3 Quantifying landscape patterns

FRAGSTATS (v4.2.598) was used to compute four different landscape pattern indices (Table 6.1) for every annual forest cover map in the sample (McGarigal *et al.*, 2012). The landscape pattern indices were only computed for forest patches. The indices were selected because they are commonly computed, provide generic, useful patterns and are comparable across scales (Haines-Young and Chopping, 1996; Kupfer, 2012). The ability of indicators to measure across scales is particularly important because the scales at which significant patterns emerge are not always obvious (Wu, 2004), and are thus largely identified through exploratory analyses. For the boreal forest, a range of scales that are potentially useful for reporting national and continental fragmentation indices remain to be tested (Wulder *et al.*, 2008d). For this study, the pattern indices defined above were calculated at four scales by varying landscape extent using a grid of landscapes with the native grain size of the Landsat imagery (0.09 ha). The scales were of approximate map extents: 50,000 ha; 25,000 ha; 10,000 ha; and 5,000 ha. To ensure that trends were only assessed for the boreal zone and that data gaps (*i.e.*, background) did not adversely affect the pattern indices, only landscapes with 25% or less background and 10% or more of the boreal zone were included in the final tabulations. Landscapes with significant agricultural influence (*i.e.*, > 1%) per the Canadian Agricultural Inventory database (Statistics Canada, n.d.) were excluded from the analysis. Core forest indices were calculated based on a 120 m depth from the edge of forest patches, which represented four Landsat pixels and was consistent with known edge effects in boreal forests (Harper *et al.*, 2005).

Autocorrelation was assessed both spatially and temporally for forest cover to provide explicit measures of spatial and temporal distances at which two landscapes could be considered independent samples for statistical analyses. Since all of the pattern indices are directly correlated with class proportion (*i.e.*, forest cover) (Remmel and Csillag, 2003), the autocorrelation of forest cover provided an indication of autocorrelation structure in the dataset overall. Latitude and longitude coordinates were calculated for the centroid of every landscape at all extents. Spatial autocorrelation was measured at each extent between the centroid coordinates of the landscapes using Moran's I index of autocorrelation (Moran, 1950). Only lag distances with at least 1,000 paired landscapes were considered in the spatial autocorrelogram. The large-lag standard error multiplied by two was computed for each lag class to assess the significance of Moran's I at $\alpha = 0.05$ for each lag distance (Anderson, 1976). Temporal autocorrelation was computed for the time series of every landscape at annual lag distances to provide an estimate of the temporal autocorrelation present in the data (Venables and Ripley, 2002). Significance of temporal autocorrelation was assessed using boxplots with interquartile range outside the 95% confidence bounds.

Temporal trends in the pattern indices were first tested for monotonicity (*i.e.*, unidirectional positive or negative change trend) using non-parametric Mann-Kendall (M-K) tests. The M-K test is robust and performs with higher power for non-normal data compared with standard parametric models (Yue and Pilon, 2004). Confidence bounds (95%) were calculated for the M-K test statistic (Tau) using a block bootstrap method with 500 repetitions in order to account for potential bias caused by autocorrelation. Theil-Sen lines were then fitted to the monotonic trends, which are robust with non-normal data and outliers compared with using standard linear regression models (Wilcox, 1998). The slope and intercept of the Theil-Sen line were calculated as the median slope of all pairwise point comparisons of the serial data. All statistical tests were performed at the significance level $\alpha = 0.05$.

Table 6.1 Landscape pattern indices that were calculated on an annual basis of forest cover.

Index	Formulation (units)	Interpretation	Relevance in boreal forests	Reference
Forest cover	Total forested area divided by total landscape area multiplied by 100 (%).	A measure of the relative coverage of forest with units that are standardized to landscape extent. Higher values indicate more forest cover relative to non-forest cover classes.	Relates to boreal forest dominance.	Lee *et al.* (2003)
Largest forest patch index	Area of the largest patch in the landscape divided by total landscape area multiplied by 100 (%).	A measure of the relative contribution of the largest forest patch to the landscape with units that are standardized to landscape extent. Higher values indicate lower fragmentation.	Relates to boreal forest intactness and forest connectivity.	McGarigal and Marks (1994)
Forest edge density	Total perimeter of forest patches divided by total landscape area (km per ha).	A measure of all forest edges in a landscape with units that are standardized to landscape extent. Higher forest edge density indicates higher fragmentation.	Relates to abundance of edge habitat and predator-prey dynamics.	James and Stuart-Smith (2000)
Core forest cover	Total core area divided by total landscape area multiplied by 100 (%).	A measure of the relative coverage of core forested area with units that are standardized to landscape extent. Higher values indicate lower fragmentation.	Relates to edge impacts on boreal forests and the relative abundance of core boreal forest habitat.	McGarigal and Marks (1994)

6.2.4 Distinguishing landscape change

I distinguished between three types of landscape pattern change for the 50,000 ha landscapes based on relative forest cover change: (1) high magnitude change; (2) moderate magnitude change; and (3) no change. The significance of M-K tests performed on the slopes of forest cover over the study period for individual 50,000 ha landscapes was used as a benchmark to determine which landscapes changed. Landscapes that did not have significant monotonic slopes in changing forest cover were considered *no change*. Landscapes that had significant monotonic slopes were partitioned into *moderate magnitude change* if the change slope was below the 90th percentile and *high magnitude change* if the change slope was above the 90th percentile.

6.3 Results

First, overall trends and emergent properties of the indices are presented for all the extents and ecozones. The results of forest recovery are introduced next. Then the results of spatial and temporal autocorrelation are presented. Finally, trends of boreal forest patterns in landscapes undergoing forest cover changes and trends of change variation among landscapes.

6.3.1 Overview

Approximately 72 Mha of the Canadian boreal zone was analyzed. Permanent water accounted for more than 8.3 Mha of the sampled boreal zone. Within the boreal zone, persisting forest and persisting non-forest (from 1985 to 2010) accounted for approximately 77% (49.2 Mha) and 18% (11.7 Mha) of the non-water area, respectively. Approximately 4% (2.8 Mha) of the non-water area or about 5% of the forested area was disturbed at some point during the study period of 1985 to 2010. In total, approximately 82% (52 Mha) of the non-water area was forested at some point during the time series.

The total number of landscapes that met the selection criteria (*i.e.*, ≤ 25% background data and ≥10% of the intersecting boreal zone) were distributed as follows across the extents of analysis: 18,185 landscapes at the 5,000 ha extent; 9,087 landscapes at the 10,000 ha extent; 3,451 landscapes at the 25,000 ha extent; and 1,662 landscapes at the 50,000 ha extent.

Spatial autocorrelation of forest cover was significant for lag distances up to approximately 600 km at the 50,000 ha extent. This distance reflects a minimum threshold that was generally greater than the distance between the sampled Landsat scenes. Forest cover was significantly correlated for up to three years as indicated by boxplots with interquartile range outside the 95% confidence interval. At temporal periods greater than three years, the autocorrelation was no greater than what would be expected by chance alone.

6.3.2 Forest recovery

Trends in recovery varied depending on the indicator and bio-climate zone. TCG and NDVI provided the fastest recovery rates while NBR and the mid-infrared spectral band provided the slowest recovery rates. The mean length of time for spectral indices to recover (reach 80% of the pre-disturbance value) was highest for NBR, 3.9 years, and lowest for TCG, 1.7 years. The cold and mesic bio-climate zone had the longest mean years to recover ranging from 1.9 years for TCG to 4.2 years for NBR, while the cool temperate and dry bio-climate zone had the shortest mean years to recover ranging from 1.6 years for TCG to 2.9 years for NBR. This result may indicate that the return of vegetation and initiation of successional processes is at least partially dependent on bio-climate zone, but the indices are also correlated. For example, the mid-infrared spectral band is a major component in the calculation of NBR and NDVI describes the contrast between the reflectances in the visible red and near-infrared spectral bands which also receive the largest weightings in the calculation of TCG. It is important to note that the number of pixels recovered is a function of number of years since disturbance. For example, only disturbances that occurred between 1987 and 1998 could have had recovery trends beyond a 10-year horizon. Therefore, the number of pixels recovered is necessarily biased towards shorter recovery times in the time series. Mean years to recover was higher for the subset of pixels that were disturbed 10 or more years ago (*i.e.*, between 1987 and 1998): 5.6 years for NBR; 4.8 years for the mid-infrared spectral band; 2.7 years for NDVI; and 1.7 years for TCG.

The majority of pixels detected as disturbed had observable trends of commencement of the return of vegetation within the first five years of disturbance regardless of the index or bio-climate zone (Figure 6.2). There was little difference between bio-climate zones for the same index with most differences observed within the first 5 years of disturbance (Figure 6.2). The mid-infrared spectral band had the greatest differences among bio-climate zones one year after disturbance, ranging from 42% for the cold and mesic bio-climate zone to 60% for the extremely cold and mesic bio-climate zone (Figure 6.2). The cumulative percentage of pixels recovered after five years was greater for NDVI (93%) and TCG (95%) compared with NBR (78%) and the mid-infrared spectral band (84%). The differences of cumulative percentage of pixels recovered were minimal after 10 years between NBR (93%), the mid-infrared spectral band (93%), NDVI (100%), and TCG (99%). These results indicate that vegetation has spectrally returned to pre-disturbance levels and that successional processes are likely to have initiated. Sites taking more time to recover may have experienced a land use change or a notably severe disturbance.

The percentage of pixels that had not recovered after five years were plotted as a function of the percentage of the pre-disturbance mean value for each index in Figure 7.3 where 80% corresponded to 80% of the mean pre-disturbance values of the recovery indicator. After five years, the majority of disturbed pixels in the extremely cold and mesic and the cold and mesic bio-climate zones for NDVI and the mid-infrared spectral band were close to meeting the 80% threshold (Figure 6.3). The trend in recovering pixels for TCG was similar to NBR (Figure 6.3), but the relative difference in cumulative recovered pixels after five years between these indices was approximately 17%. Disturbed pixels in the extremely cold and mesic and cold and mesic bio-climate zones had higher rates of recovery compared with the cool temperate and dry bio-climate zones (Figure 6.3).

The mid-infrared spectral band and NBR have been shown to be related to structural components of vegetation while NDVI and TCG are more sensitive to levels of chlorophyll (Cuevas-González *et al.*, 2009; Epting and Verbyla, 2005). Both the mid-infrared spectral band and NBR are known to represent the recovery of non-leafy structural attributes of vegetation beyond the initial flush of grasses, herbs and shrubs (Schroeder *et al.*, 2011; Wulder, 1998). The curves of non-recovered pixels for the mid-infrared band and NDVI likely represent variation of

Figure 6.2 Percentage of pixels recovered by the number of years following disturbance for each index and bio-climate zone. The number of disturbed pixels is equal for all indices.

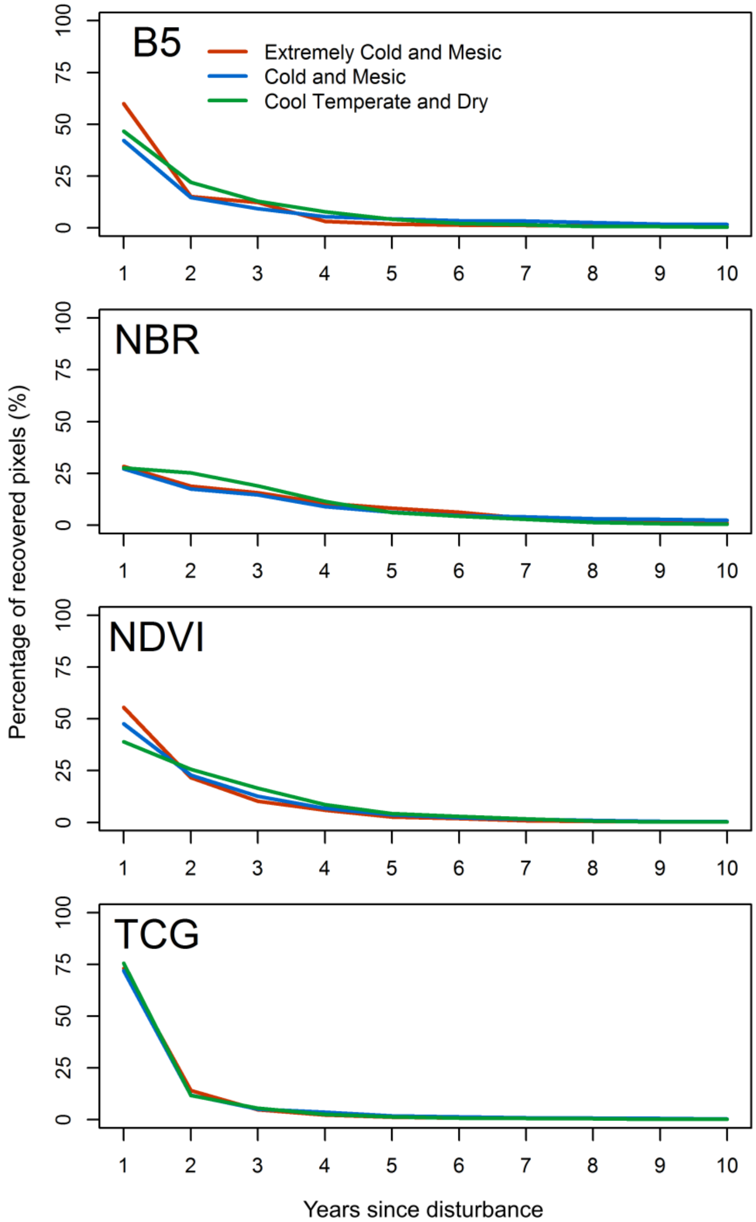
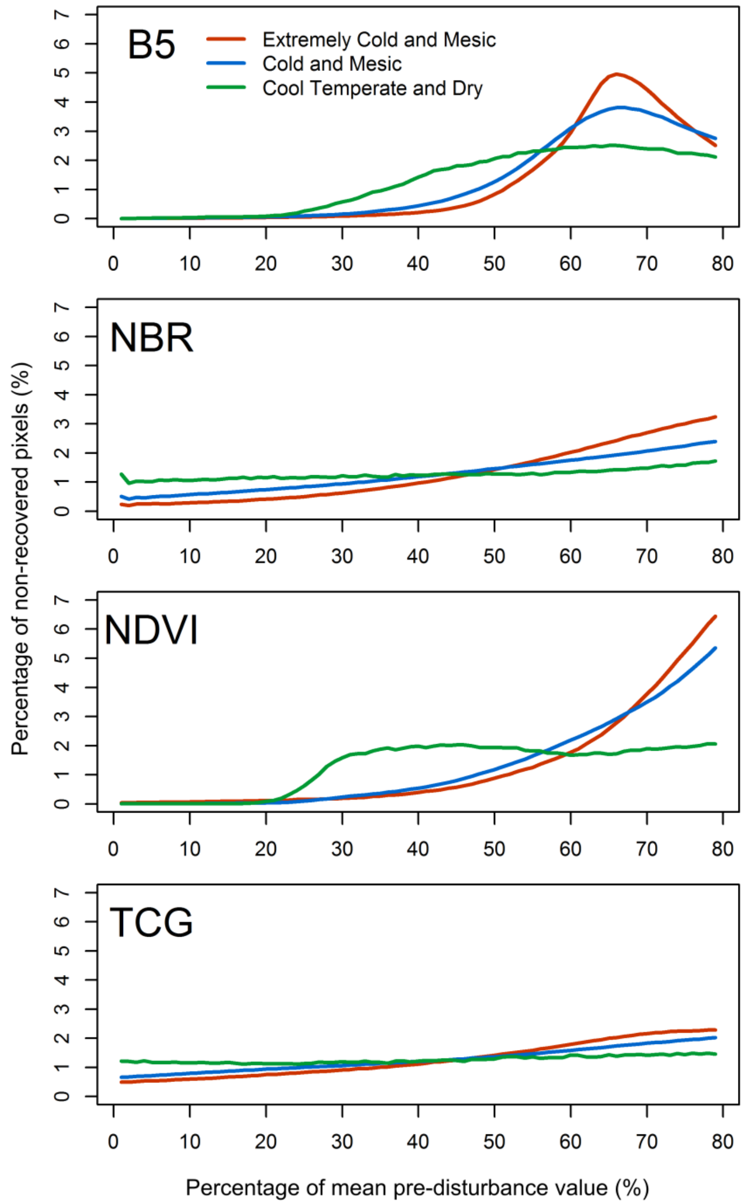


Figure 6.3 Percentage of non-recovered pixels by percentage of the mean pre-disturbance value for each index and bio-climate zone investigated after five years.

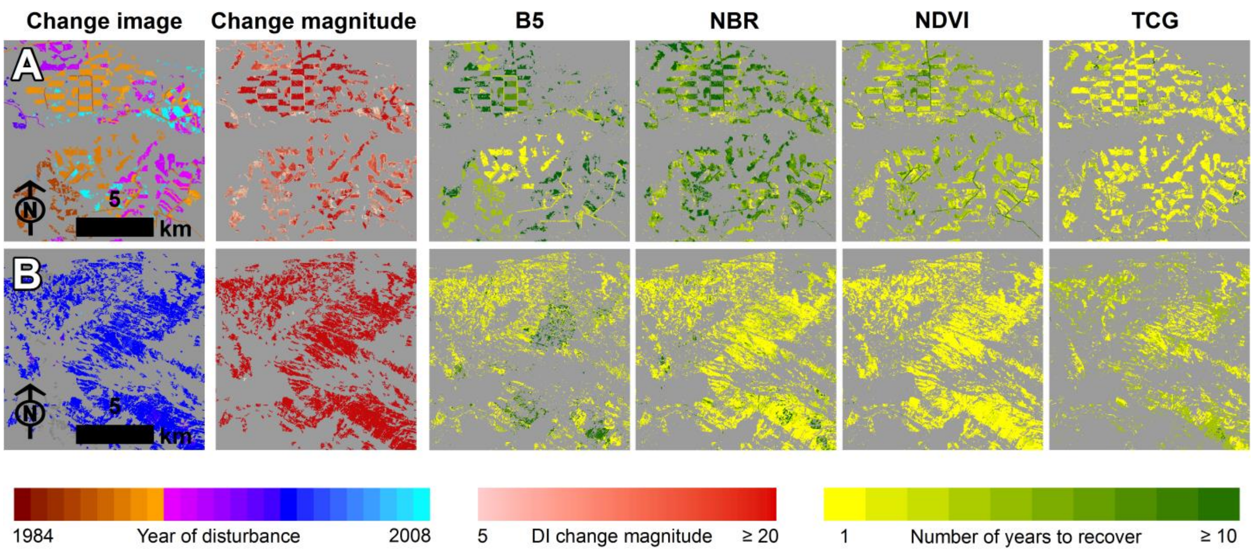


spectral recovery across a gradient of disturbance severity (Figure 6.3). By contrast, the linear curves of NBR and TCG are likely an artifact of continually increasing near-infrared reflectance as leaf cover accumulates within the disturbed pixel (Figure 6.3). Thus, the results indicate that the mid-infrared spectral band and NBR are expected to be more suitable for tracking vegetation succession following disturbance when compared to NDVI and TCG. Based on knowledge of successional processes (Oliver, 1981), the signal from visible and near-infrared wavelengths is expected to quickly saturate following disturbance due to the rapid colonization of early pioneer and shade-intolerant species.

Recovery rates among the bio-climate zones suggest that there may be subtle spectral differences in successional processes. For example, NDVI and the mid-infrared spectral band were best suited for observing differences between the bio-climate zones (Figure 6.2). The higher rate of recovery observed for the extremely cold and mesic bio-climate zone in the first year post-disturbance could be related with lower pre-disturbance mean values compared to the other zones when using the mid-infrared spectral band and NDVI. Trends in NBR recovery were much more gradual while TCG saturated one year after disturbance (Figure 6.2). Recovery rates among the bio-climate zones during later succession (*i.e.*, ≥ five years post-disturbance) were also best observed using NDVI and the mid-infrared spectral band (Figure 6.3). There was a notable lag effect for recovery of disturbed pixels in the cool temperate and dry bio-climate zone after five years with the mid-infrared spectral band and NDVI indices compared with the other bio-climate zones (Figure 6.3). The frequency distribution of non-recovered pixels in the cool temperate and dry bio-climate zone suggests that these forests take longer to recover (Figure 6.2) and after five years may still be far from reaching the recovery definition with the mid-infrared spectral band and NDVI (Figure 6.3).

The results should be interpreted with some caution because recovery was not distinguished for different disturbance types. It was evident that the choice of recovery indicator was important for different types of disturbance. For example, the initial flush of leafy vegetation following a fire or harvest saturates the recovery signal for NDVI and TCG. Additionally, disturbance legacies are evident in the recovery rates from visual assessment of harvesting and fire (Figure 6.4). For example, roads within harvests appear to recover more slowly than the surrounding forest using NDVI compared with TCG (Figure 6.4a). Recovery is remarkably

Figure 6.4 Number of years to recover by index for examples of forest harvesting in Alberta (A, path45/row 23, 117° 17' W 53° 44' N) and a fire in Northwest Territories, Canada (B, path57/row 14, 128° 12' W 65° 33' N). The year of disturbance color ramp applies to the change image, the disturbance index (DI) change magnitude color ramp applies to the change magnitude image, and the number of years to recover color ramp applies to the recovery images.



heterogeneous within fires, this is primarily evident with the mid-infrared spectral band or NBR (Figure 6.4b). The examples shown in Figure 6.4 are typical of forest management occurring in the southern Canadian boreal forest and fire which is much more common in occurrence and extent throughout the northern Canadian boreal forest.

Using the disturbance maps produced by the VCT, forest recovery was defined as 80% of the mean Normalized Burn Ratio value from the two years before the disturbance. The Normalized Burn Ratio was selected because it is a robust recovery indicator compared to other spectral indices (Pickell *et al.*, 2016) and likely represents the structural components of vegetation (Banskota *et al.*, 2014; Pflugmacher *et al.*, 2013).

The NBR index was selected for mapping the forest cover gains over the time series in support of the fragmentation analysis. The NBR index was more robust within the ecozones and provided a more conservative view of forest recovery with mean recovery years similar to what would be expected for the Canadian boreal forest.

6.3.3 Forest cover

The majority of landscapes at all scales had significant forest cover change. Approximately 74% (1,232) of 50,000 ha landscapes were characterized by moderate magnitude change and 8% (134) were characterized by high magnitude change. The mode of forest cover for the 50,000 ha extent shifted over the observation period from approximately 85% in 1985 to 79% in 2010 (Figure 6.5). The median value of forest cover ranged from approximately 73% in 1985 to 70% in 2010 for the 50,000 ha extent and from 77% in 1985 to 74% in 2010 for the 5,000 ha extent. The Boreal Shield, Hudson Plain, Boreal Plain, Montane Cordillera, Taiga Plain, and Boreal Cordillera ecozones were the most forested with median forest cover greater than 50%. There was a significant negative trend over time in forest cover across most ecozones and extents (Figure 6.6). The most forested ecozones underwent the largest magnitude changes in forest cover in terms of the median Theil-Sen slope (Figure 6.6). The Montane Cordillera ecozone was the only ecozone that did not have a significant negative trend in forest cover (Figure 6.6) despite being among the most forested ecozones. The Boreal Shield, Boreal Plain, and Boreal Cordillera ecozones had the greatest median forest cover change over the study period in excess of -0.1% ·

Figure 6.5 Histograms of each index for the 50,000 ha landscape extent from 1985 to 2010.

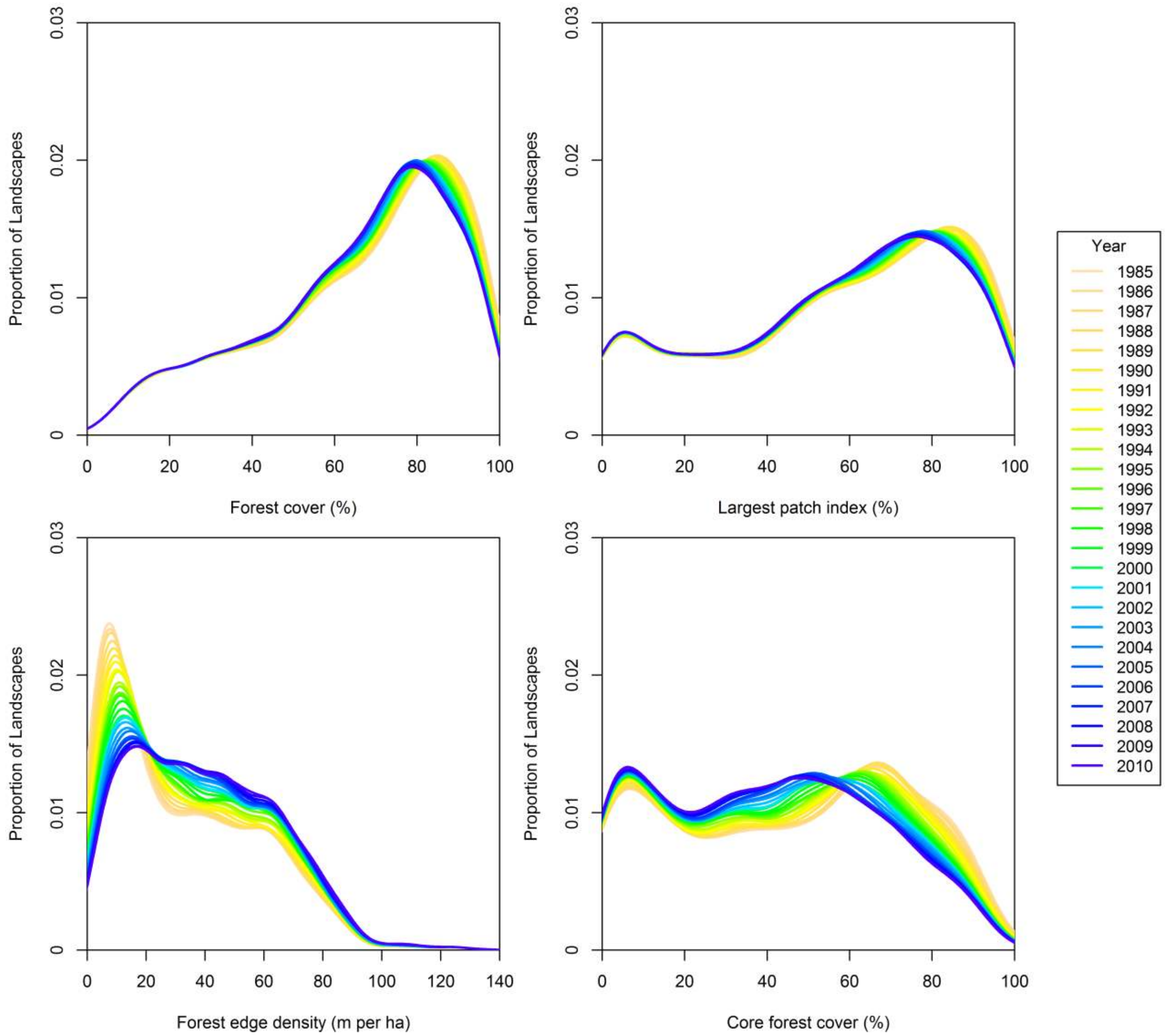
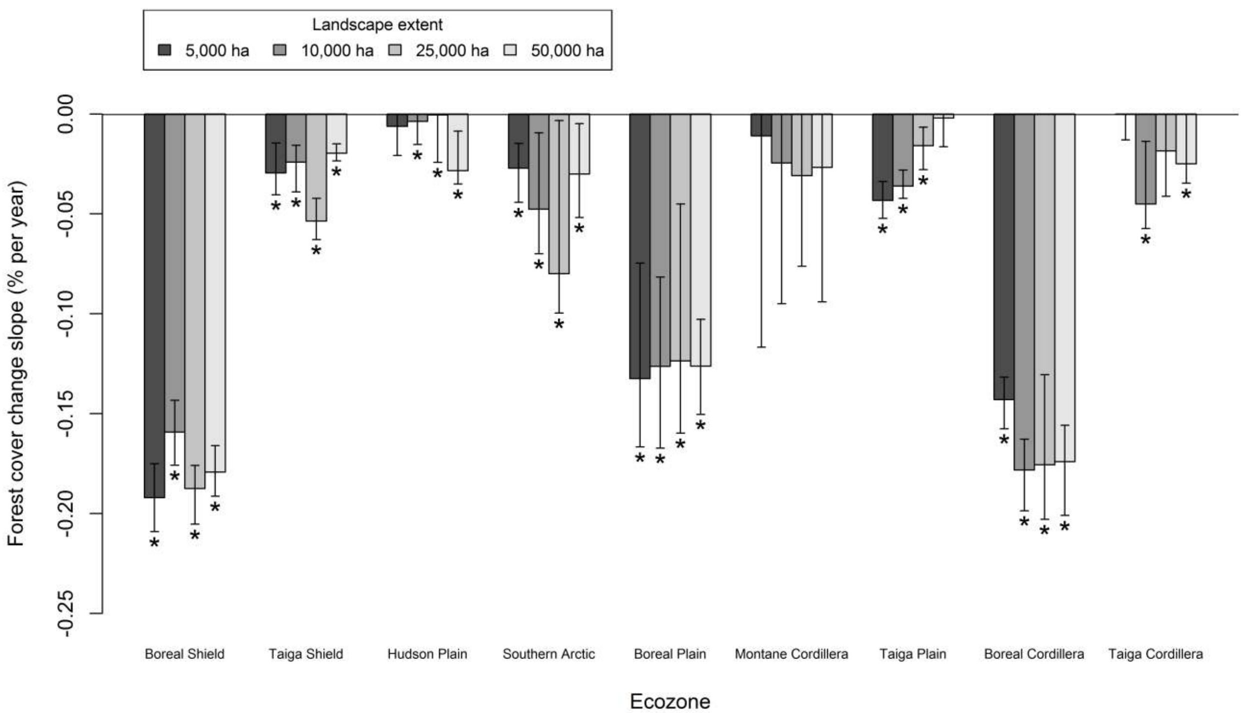


Figure 6.6 Median Theil-Sen slope of forest cover from 1985 to 2010 for each ecozone and landscape extent. Error bars indicate the upper and lower 95% confidence bounds. Asterisks (*) indicate significant ($p < 0.05$) monotonic trends from a Mann-Kendall test.



yr^{-1} (Figure 6.6). Landscape extent had significant effects on observations of forest cover decline for the Boreal Shield, Taiga Shield, Taiga Plain, and Boreal Cordillera ecozones (Figure 6.6).

6.3.4 Forest patch size

The largest forest patch index declined significantly over the study period for all landscape extents and ecozones (Figure 6.7). The greatest declines in the largest forest patch index were observed for the Boreal Shield, Boreal Plain, and Boreal Cordillera ecozones (Figure 6.7). Landscape extent was associated with the observed change rate for the Boreal Shield, Taiga Plain, and Boreal Cordillera ecozones (Figure 6.7). The Boreal Plain ecozone had the greatest median value of largest forest patch index (80%) and the Taiga Cordillera ecozone had the lowest median value (16%) in 1985. The mode of largest forest patch index shifted over the study period from approximately 85% in 1985 to 76% in 2010 (Figure 6.5). The median value of largest forest patch index declined by approximately -0.09% $\cdot$ yr^{-1} for all landscapes. Declining largest forest patch was associated with the declining trend in forest cover.

6.3.5 Forest edge

Forest edge density increased for all landscape extents and ecozones over the study period (Figure 6.8). The median value of forest edge density increased at a rate of approximately 0.5 m $\cdot$ ha^{-1} $\cdot$ yr^{-1} from 24 m $\cdot$ ha^{-1} to 38 m $\cdot$ ha^{-1} for all 50,000 ha landscapes. The most forested ecozones underwent the largest magnitude changes in forest edge density for 50,000 ha landscapes: 0.8 m $\cdot$ ha^{-1} $\cdot$ yr^{-1} in the Boreal Shield; 0.5 m $\cdot$ ha^{-1} $\cdot$ yr^{-1} in the Boreal Plain; and 0.4 m $\cdot$ ha^{-1} $\cdot$ yr^{-1} in the Boreal Cordillera ecozone (Figure 6.8). Landscape extent had a significantly affected the Boreal Shield and Taiga Plain Ecozones (Figure 6.8). The number of landscapes with low forest edge density (*i.e.*, < 25 m $\cdot$ ha^{-1}) declined while landscapes with intermediate levels of edge density increased over the study period for the 50,000 ha landscapes (Figure 6.5).

Figure 6.7 Median Theil-Sen slope of largest forest patch index from 1985 to 2010 for each ecozone and landscape extent. Error bars indicate the upper and lower 95% confidence bounds. Asterisks (*) indicate significant (p < 0.05) monotonic trends from a Mann-Kendall test.

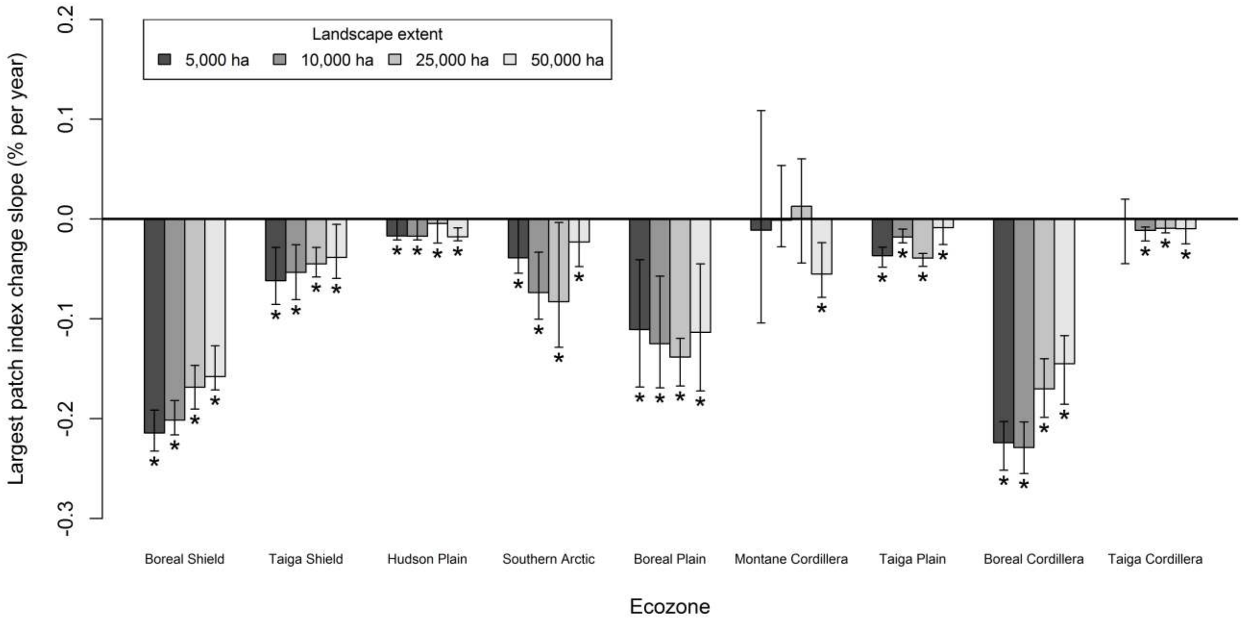
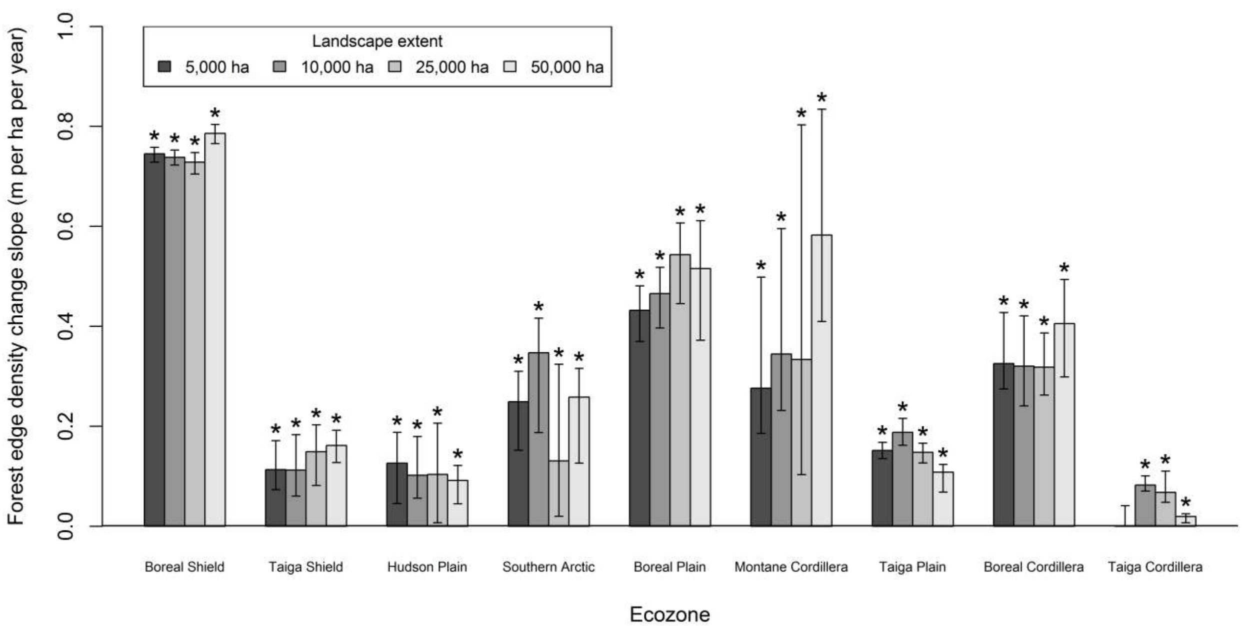


Figure 6.8 Median Theil-Sen slope of forest edge density from 1985 to 2010 for each ecozone and landscape extent. Error bars indicate the upper and lower 95% confidence bounds. Asterisks (*) indicate significant (p < 0.05) monotonic trends from a Mann-Kendall test.



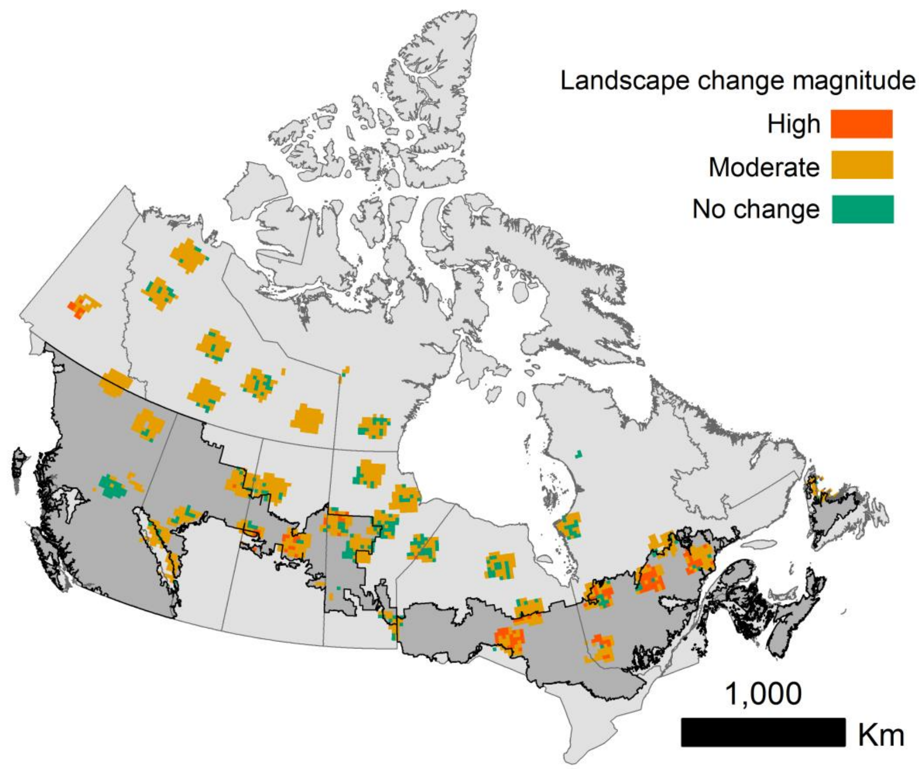
6.3.6 Core forest cover

Core forest cover declined significantly for all landscape extents and ecozones and this trend was strongly associated with the trend of declining forest cover in general and with the declining largest forest patch index. The Boreal Shield, Boreal Plain, and Boreal Cordillera ecozones had the largest magnitude changes in core forest cover per year. Generally, landscape extent had little effect on the observed change rates of core forest cover within ecozones. The number of 50,000 ha landscapes with high core forest cover (*i.e.*, > 70%) declined over the study period by 41% from 413 in 1985 to 243 in 2010 (Figure 6.5). Within the ecozones, core forest cover of the 50,000 ha landscapes declined significantly at a rate of approximately -0.6% $\cdot$ yr^{-1} in the Boreal Shield, -0.5% $\cdot$ yr^{-1} in the Boreal Plain, -0.1% $\cdot$ yr^{-1} in the Montane Cordillera, -0.1% $\cdot$ yr^{-1} in the Taiga Plain, and -0.2% $\cdot$ yr^{-1} in the Boreal Cordillera. The rate of core forest decline in the Boreal Shield (-0.6% $\cdot$ yr^{-1}) was significantly different than all other ecozones except the Boreal Plain ecozone. Core forest cover was bi-modal for the 50,000 ha extent in 1985 with modes occurring at approximately 6% and 67% cover (Figure 6.5).

6.3.7 High magnitude changes in boreal forest landscapes

In total, 166 (10%) of the 50,000 ha landscapes were identified as high magnitude change, which were those ranked above the 90th percentile in terms of forest cover change slope. These rapidly changing landscapes were primarily distributed within the managed areas of the boreal forest, defined as forested lands tenured for wood production. Some landscapes in the Yukon were also high magnitude changes due to active fire years during the observation period (Figure 6.9). Within the ecozones, 126 (21%) of the landscapes in the Boreal Shield were high magnitude changes, 27 (12%) in the Boreal Plain, and 13 (16%) in the Boreal Cordillera. The median value of largest forest patch index declined by approximately -0.4% $\cdot$ yr^{-1} from 83% to 73%, for high magnitude change landscapes, which was four times greater than all landscapes. Also, the median value of forest edge density increased by approximately 1.4 m $\cdot$ ha^{-1} $\cdot$ yr^{-1} from 10 m $\cdot$ ha^{-1} to 45 m $\cdot$ ha^{-1} for high magnitude change landscapes and core forest cover declined by approximately -1.1% $\cdot$ yr^{-1} from 69% to 42%. The landscapes that underwent large changes in

Figure 6.9 Locations of 50,000 ha landscapes with landscape change magnitude relative to boreal forests managed for wood production (dark gray).



terms of the pattern indices were among the most forested initially, with median forest cover approximately equal to the 75^{th} percentile of forest cover for all landscapes in 1985.

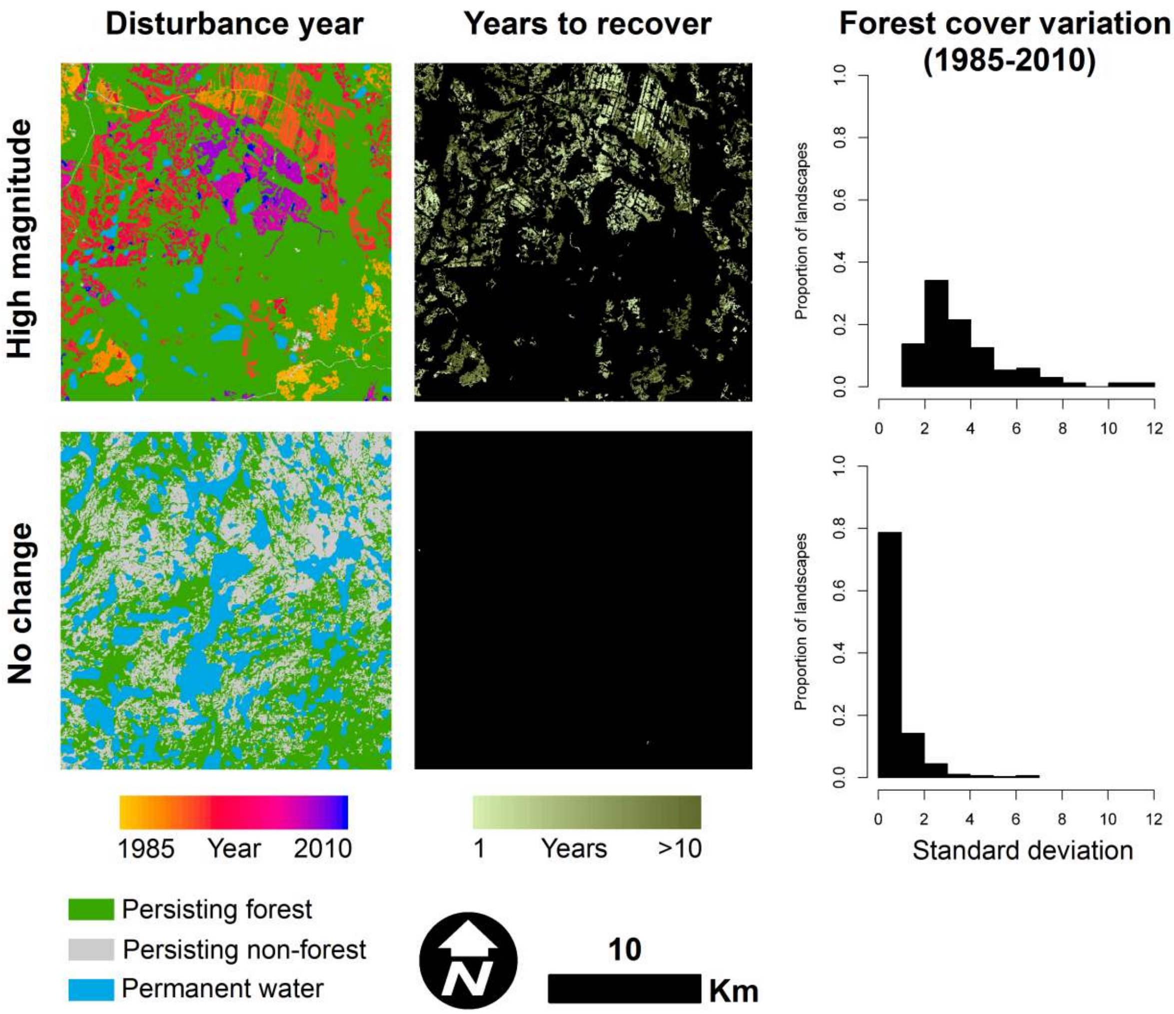
6.3.8 Boreal forest pattern variation

Approximately 18% of all 50,000 ha landscapes did not change due to low forest cover change slopes (*e.g.*, less than $|0.01\%| \cdot yr^{-1}$). These no change landscapes were distributed as follows among the ecozones (percentage expressed out of total landscapes in that zone): 100 (18%) in the Boreal Shield; 54 (20%) in the Taiga Shield; 33 (16%) in the Boreal Plain; 30 (41%) in the Montane Cordillera; 56 (43%) in the Hudson Plain; 29 (14%) in the Taiga Plain; eight (15%) in the Southern Artic; and one (33%) in the Taiga Cordillera. On average, these relatively stable landscapes were approximately 67% forested ($\tilde{X}$ = 71%) initially in 1985 with 12% of the landscape comprised of permanent water ($\tilde{X}$ = 8%). The median value of the largest patch index declined at a rate of -0.06% $\cdot$ yr^{-1} from 65% to 63%. Forest edge density increased by approximately 0.4 m $\cdot$ ha^{-1} $\cdot$ yr^{-1} from 27 m $\cdot$ ha^{-1} to 37 m $\cdot$ ha^{-1}. Core forest cover declined at a rate of approximately -0.2% $\cdot$ yr^{-1} from 47% to 41%. A comparison between examples of high magnitude forest pattern change and no change is shown in Figure 6.10. High magnitude change landscapes were associated with increased forest cover variation over time relative to no change landscapes (Figure 6.10).

6.4 Discussion

The results presented here represent a first look at intrinsic and extrinsic landscape patterns of boreal forests across a sample of the available satellite archive. I provide some answers to the topical questions originally put forward by Cardille *et al.* (2005) using Canadian boreal forests as a case study.

Figure 6.10 Examples of high magnitude change landscape pattern in central Ontario (upper images) and no change landscape pattern in Northwest Territories (lower images). The panels on the right refer to the forest cover variation, as measured by the standard deviation, over the time series for all 50,000 ha landscapes in the high magnitude change and no change landscape classes.



6.4.1 How have boreal forest spatial patterns changed over time?

The temporal dimension of the pattern database allowed me to investigate how spatial patterns of boreal forests have changed over time. Generally, landscapes were characterized by moderate declines in forest cover and this was associated with declining largest patch size and core forest cover, and increasing forest edge density. Forest cover declined by approximately -0.06% $\cdot$ yr^{-1} on average, which was consistent with rates of high magnitude, stand-replacing disturbance in the boreal forest (Hansen *et al.*, 2013). The high magnitude change landscapes were also more likely to be the most forested landscapes. This result suggests that there is a direct relationship between the amount of forest cover and the likelihood and degree of change.

Although I examined different landscape extents, the results also agree with the findings of Wulder and colleagues (2008d) in terms of how the pattern indices vary by ecozone. For example, I found that forest cover generally increased as landscape extent decreased in all ecozones except the Taiga Cordillera ecozone (Wulder *et al.*, 2008d). Moreover, I found a similar association between ecozone and forest edge density. The least forested ecozones (Taiga Cordillera, Taiga Plain, and Taiga Shield) had the lowest levels of forest edge density relative to the other ecozones *circa* 2000 (Wulder *et al.*, 2008d). The results of the time series pattern database suggest that changes in forest patterns varied significantly among ecozones over the last three decades. This could be due to broad-scale climatic influences on boreal forests that are represented by the ecozones such as precipitation and temperature.

6.4.2 What are the drivers of spatial boreal forest patterns?

The general change trends noted here are consistent with what is known about resource development in boreal forests of Canada. Forests managed for wood production were associated with high magnitude landscape pattern changes. For example, the largest declines in forest cover were observed for the Boreal Shield, Boreal Plain, and Boreal Cordillera ecozones. These are regions that are known to be impacted by intensive forest management and energy development (Bergeron *et al.*, 2007; Pickell *et al.*, 2013; Schneider *et al.*, 2003). The forest patterns of the eastern landscapes of central Ontario and southern Québec can be attributed to progressive clear cutting of the forest between 1985 and 2010 (Bouchard and Pothier, 2011). Similarly, the forest patterns of the western landscapes of Alberta, Saskatchewan, and Manitoba can be attributed to road construction, forest harvesting, and energy development around the oil sands and the foothills of the Rocky Mountains (Pickell et al., 2015). The Boreal Shield ecozone had the fewest landscapes that did not change over the study period relative to the total number of landscapes sampled in that zone (17%). By contrast, 76% of all high magnitude change landscapes occurred in the Boreal Shield, which represented 21% of all landscapes sampled in that zone. The significance of this finding should be underscored in terms of the relative importance of this ecozone to the Canadian forestry industry, which accounts for approximately one-third of all wood volume in Canada (Canada Forest Inventory, 2013; Gillis *et al.*, 2005). These results suggest that the impacts of forest management are not evenly distributed across the boreal forest, but rather have been focused on the most forested landscapes, typically in the southern Canadian boreal forest. The ecological implications of this include disproportionate impacts on forest-dwelling species that rely on large, contiguous patches of forest.

Landscapes that did not change were associated with minimal disturbance, were approximately two-thirds forested, and were characterized by less than 40 $m \cdot ha^{-1}$ forest edge density on average. Core forest cover and the largest forest patch comprised approximately 47% and 65%, respectively, of landscape area at the median in these landscapes in 1985 and the annual change of forest cover was relatively small (less than $0.01\% \cdot yr^{-1}$ on average). These results indicate that nearly one-fifth (18%) of all 50,000 ha landscapes are relatively intact, yet still undergo small changes in forest cover, probably due to localized disturbance.

Carpenter and Brock (2006) have suggested that rising variance is an indicator of transition. I found no evidence for rising variance in forest cover in the time series, however, the standard deviation of edge density nearly doubled for the high magnitude change landscapes. The variance of forest edge density for all 50,000 ha landscapes over the time series was otherwise stable. These results suggest that managed Canadian boreal forest landscapes are being driven beyond the present range-of-variation of landscape structure. Other studies have linked negative outcomes on managed boreal forest landscapes with rising edge density such as increased road networks and habitat attenuation (Franklin and Forman, 1987; Lesmerises *et al.*, 2013, 2012).

6.4.3 What is the historical context for assessing present-day boreal forest spatial patterns?

The pattern database results provide important context for forest management agencies committed to implementing ecosystem-based management (EBM) strategies. The specific trends noted over the last three decades demonstrate that the rate of high magnitude forest cover change in boreal forests tenured for wood production is generally inconsistent with other regions of the Canadian boreal forest. There was sufficient evidence to demonstrate that the magnitude of changes in the pattern indices was significantly greater for managed landscapes relative to unmanaged landscapes. The 26-year observation period was sufficient to observe multiple harvest passes within the managed landscapes. The results suggest that three decades of “business-as-usual” forest harvesting have not produced the desired landscape patterns relative to unmanaged boreal forests despite the fact that many jurisdictions across Canada have adopted EBM harvesting guidelines (Delong and Tanner, 1996; FSC, 2004; Klenk *et al.*, 2008; Ontario Ministry of Natural Resources, 2001). The relatively recent adoption of these management strategies (*e.g.*, post-2000) makes it especially challenging to observe outcomes relative to the observation period, particularly for landscapes in Ontario and Québec where the majority of rapidly changing landscapes were observed. The implementation of EBM principles continues to be in a transition period and measureable outcomes in cumulative landscape patterns may not be observable for several more decades.

Chapter 7

Conclusions

7.1 Research innovations

This dissertation provided key innovations for the analysis of anthropogenic patterns from event to landscape scales:

- A reconceptualization of anthropogenic disturbance regimes in Chapter 2 provided new insights into the significant impacts of human resource appropriation. The new knowledge can support research priorities and planning initiatives at the landscape level as in Chapters 3 and 4.
- Anthropogenic disturbance event patterns were compared to fire burn patterns for the first time to the best of my knowledge in Chapters 3 and 4. The insights generated from this novel approach can reshape how resource extraction activities can be planned from local to regional scales.
- An innovative approach to integrate disturbance and recovery models was developed for pattern analysis of boreal forested landscape patterns in Chapters 5 and 6.
- Temporal landscape patterns of boreal forests were analyzed across broad scales in Chapter 6. The results significantly advance our understanding of changing landscape patterns over the recent three decades for the Canadian boreal forest.

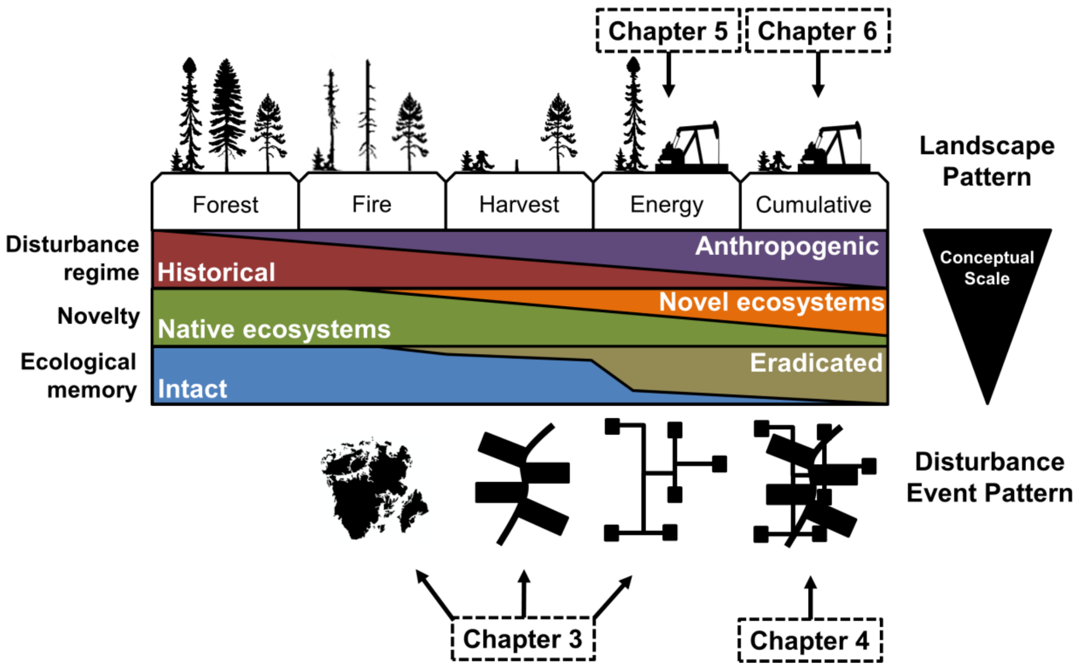
7.2 Progress towards characterization of anthropogenic disturbance

Anthropogenic disturbance regimes are complex and dynamic phenomena. They exhibit characteristics at multiple spatial and temporal scales that are often associated with other socio-economic phenomena such as resource development (Masek *et al.*, 2013). Observation and characterization of more complete anthropogenic disturbance regimes that incorporate forest management and energy development of forests will become requisites for understanding the consequences of forest use in the Anthropocene (Gauthier *et al.*, 2015). The substantial increase in energy development and consumption over recent decades, in particular fossil fuels, will likely undermine ecosystem-based management (EBM) of the Canadian boreal forest (Schneider *et al.*, 2003). Ecological consequences such as biodiversity and habitat loss have already been exacerbated by energy development (Butt *et al.*, 2013) and these activities are likely to eradicate ecological memory of the boreal forest (Figure 7.1).

I determined in Chapter 3 that forest management activities better approximated the historical range-of-variability (HRV) compared to the disturbance activities of the energy sector. Attempts to align harvest designs with historical fire patterns via aggregate harvesting patterns since 2000 was successful relative to traditional two-pass dispersed harvesting. In particular, I observed that the level of residuals of aggregate harvests were statistically no different from the HRV. These findings have significant ramifications for the maintenance of boreal forest resilience where ecological memory is largely sustained from residual and legacy structures following disturbance (Bengtsson *et al.*, 2003; Drever *et al.*, 2006; MacFarlane, 2003). Moreover, the results suggest that ideal disturbance patterns desired under EBM cannot be achieved through the efforts of individual resource sectors. I demonstrated in Chapter 4 that landscapes impacted by multiple resource sectors can lead to novel ecosystem spatial patterns (Figure 7.1). Therefore, a concerted effort to better understand and manage for the cumulative effects of anthropogenic disturbance patterns is required if EBM is to be successful in boreal forests (Schneider *et al.*, 2003).

In general, anthropogenic disturbance regimes in Canada's boreal forest have been largely driven by forest harvesting (Gauthier *et al.*, 2015). However, I showed in Chapter 4 that the development of energy resources in the western Canadian boreal forest since 2000 has significantly shifted anthropogenic disturbance regimes in terms of extent, severity, and persistence. The transition of the anthropogenic disturbance regime following the significant increase in energy development was marked by overall reductions in remnant levels and patch size. Generally, cumulative anthropogenic disturbances continued to shift further away from the HRV over the last decade. Some of the cumulative anthropogenic disturbance patterns were regionally-dependent, likely stemming from differences in management practices and available resources. The findings suggest that implementation of EBM practices cannot be generalized over large areas and must be adapted to local and regional conditions.

Figure 7.1 Revisiting the conceptual framework that was developed in Chapter 2 and links the results of the dissertation.



I demonstrated in Chapter 3 that energy resource development has shifted land use priorities from forest management towards non-renewable resources and the consequence has been a disjunctive land planning process (Schneider *et al.*, 2003). For example, energy resource extraction is planned separately from forest harvesting (Schneider *et al.*, 2003). Thus, many of the cumulative impacts from the transitioning anthropogenic disturbance regime are poorly understood. New concerns have emerged around the decommissioning and reclamation of well sites, seismic lines, and mines (Lee and Boutin, 2006; MacFarlane, 2003, 1999; Osko and Macfarlane, 2001; Simmons *et al.*, 2008), which could impact the resilience of the Canadian boreal forest in terms of ecological memory (Drever *et al.*, 2006; Figure 7.1).

Historical range-of-variability analysis is a robust analytical method that can provide meaningful insights into ecological patterns (Landres *et al.*, 1999; Morgan *et al.*, 1994). Chapters 2 and 3 made significant contributions towards our understanding of the characterization of anthropogenic disturbance regimes relative to the HRV. Investigating land use policy changes can be challenging to assess without some understanding of the range-of-variability of the anthropogenic disturbance patterns during different periods of major resource development (Masek *et al.*, 2013; Nonaka and Spies, 2005).

7.3 Progress towards detection of anthropogenic disturbance

The satellite spectral time series approach in Chapter 5 was able to identify both fires and resource extraction disturbances in the southwestern Canadian boreal forest. The year of disturbance was recorded within one year of the forest inventory date for approximately 89% of disturbed forest by area. The attribution of disturbance type was accomplished with approximately 94% overall accuracy. The classification of disturbance allowed for analysis of source disturbance rates, which determined that resource development was the primary driver of vegetation cover change. This finding in Chapter 5 confirmed that fire suppression was associated with high levels of anthropogenic disturbance (Figure 7.1). I foresee satellite time series approaches being utilized to map historical forest disturbances and track the trends of human appropriation of forested ecosystems. Such information may help to predict rates of change in forested landscapes that may undergo significant resource development in the future.

Knowledge of historical anthropogenic disturbance trends can significantly improve landscape planning and forest management, particularly where biodiversity and habitat are rapidly declining due to anthropogenic land cover changes. Specifically, the disturbance year attribution by the vegetation change tracker (VCT) may improve or validate stand origin estimation in standard forest inventory attribution. Additionally, the combination of the change detection procedure and the disturbance type classification can be used to automate feature extraction of roads or well sites. This is a robust approach for examining historical disturbance trends related to human modification of the environment.

7.4 Progress towards characterization of boreal forest landscape structure

The integrated pattern analysis and forest cover change detection method in Chapter 6 improves our capacity to observe the impacts of standing-replacing disturbance on boreal forest patterns. The results demonstrate that the indices were robust at observing changes to boreal forest patterns at a range of spatial scales. I focused primarily on the forest patch area and edge indices due to their intuitive interpretation and relevance to boreal forest ecosystems. The pattern analysis demonstrated that indices standardized to landscape extent (*i.e.*, scale) were well-suited to a cross-scale analysis of spatial patterns (Wu, 2004).

The long term temporal information of the pattern database developed in Chapter 6 provided an opportunity to assess how landscape patterns of boreal forest cover changed in response to drivers of forest cover change. The method was useful for estimating changes to forested landscape structure over broad spatial and temporal scales. I demonstrated in Chapter 6 that changes in some indices, such as edge density and core forest cover, may reflect the emergence of novel spatial patterns for boreal forests (Figure 7.1). The nature of the pattern results was consistent with what is known about resource development patterns in the Canadian boreal forest. The largest proportion of high magnitude change landscapes was within boreal forests managed for wood production. Additionally, inherent variability in forest cover due to terrain and localized disturbance was observed for some landscapes, which are relatively intact. The results suggest that the landscape pattern indicators were sufficiently sensitive to even small changes in forest cover. Databases like this provide range-of-variability measures that can aid future research in contextualizing the configuration and composition of boreal forested landscapes in space and time.

7.5 Synthesis of metrics used to assess anthropogenic disturbance regimes

A wide range of metrics were used in this dissertation to characterize disturbance and landscape patterns resulting from anthropogenic disturbance regimes. Land managers and governments require metrics that are transferrable, repeatable, and convey important information about ecological condition (Canadian Forest Service, 1995). The pattern metrics used in this dissertation were selected on the basis of these conditions. Table 7.1 summarizes the usefulness of the pattern metrics by scale and ecological relevance.

Metrics computed at the disturbance event scale represent opportunities for forest licensees to evaluate their success towards emulating the historical fire regime. For example, metrics describing the historical range-of-variability of undisturbed remnant forest following fire can significantly improve the planning of forest harvests by including a simple percentage benchmark into harvest designs (Dzus *et al.*, 2009). By contrast, metrics computed at the landscape scale are probably of more general interest to land managers and government regulators who seek to better understand pattern outcomes from the activities of multiple stakeholders within their jurisdiction. For example, forest edge density and core forest cover may be critical indicators for regulating human land use activities within the endangered caribou range of Alberta (Dyer *et al.*, 2001). Thus, the relevant scale at which a metric is applied is crucial for interpreting patterns of anthropogenic appropriation of boreal forests. For example, the largest disturbed patch was critical for identifying differences between anthropogenic disturbance events and the historical fire events in chapters 2 and 3, but a similar metric of largest forested patch applied at the landscape scale in chapter 6 had little bearing on overall trends and may have been redundant with forest cover.

Table 7.1 Summary of metrics used to characterize anthropogenic disturbance regimes in the Canadian boreal forest.

Pattern metric	Useful scale	Ecological relevance	Reference
Disturbance event area	Disturbance event	Total area of influence of a disturbance event	Burton *et al.* (2008), Andison (2012)
Mean disturbed patch area	Disturbance event	Relates to the abundance of regenerating forest in a disturbance event	McGarigal and Marks (1994)
Number of disturbed patches	Disturbance event	Patchiness of a disturbance event	Andison and McCleary (2014)
Largest disturbed patch area	Disturbance event	Relates to disturbance severity and disturbance dominance	Cumming (2001a)
Mean island remnant area	Disturbance event	Relates to forest structure and sources of ecological memory	Bengtsson *et al.* (2003)
Number of island remnants	Disturbance event	Relates to the number of potential seed sources in a disturbance event	Andison and McCleary (2014)
Largest island remnant area	Disturbance event	Dominance of undisturbed remnant forest in a disturbance event	Andison and McCleary (2014)
Percentage of island remnants	Disturbance event	Relative abundance of isolated undisturbed remnant forest in a disturbance event	Andison and McCleary (2014)
Percentage of matrix remnants	Disturbance event	Relative abundance of undisturbed remnant corridors in a disturbance event	Andison and McCleary (2014)
Percentage of total remnants	Disturbance event	Relative abundance of structures that might represent ecological memory in a disturbance event	Andison and McCleary (2014)
Percentage of largest disturbed patch	Disturbance event and landscape	Relative dominance of disturbed forest	Andison and McCleary (2014)
Disturbance and forest edge density	Disturbance event and landscape	Relates to abundance of edge habitat and predator-prey dynamics	James and Stuart-Smith (2000)
Forest cover	Landscape	Relates to boreal forest dominance	Lee *et al.* (2003)
Largest forest patch index	Landscape	Relates to boreal forest intactness	McGarigal and Marks (1994)
Core forest cover	Landscape	Relates to edge impacts on boreal forests and the relative abundance of core boreal forest habitat	McGarigal and Marks (1994)

7.6 Research limitations

The output from the NEPTUNE (Novel Emulation Pattern Tool for Understanding Natural Events) model requires careful consideration of the underlying assumptions used to map disturbance events. For example, varying the spatial buffer and recovery times used to map disturbance events can result in patterns different from what I observed. The 200 m spatial buffer used was derived from research on fires and used to make the anthropogenic disturbance patterns comparable to the HRV. Varying the spatial buffer can change the size of disturbance events and the size distribution of patches within events (Andison, 2012), although it would also invalidate any comparison to the HRV.

Similarly, varying the recovery times used for different disturbance types could change disturbance pattern outcomes, especially related to remnant levels and edge density. For example, roads and seismic lines that are allowed to persist in the model will slightly lower overall remnant levels, but will significantly convert matrix remnants to island remnants and increase edge density of disturbance events. Moreover, disturbed features are assumed to recover at a constant rate despite known variability in the recovery times of seismic lines (Lee and Boutin, 2006; MacFarlane, 2003; Osko and Macfarlane, 2001; Van Rensen, 2014), well sites (MacFarlane, 1999), and harvests (Beckingham and Archibald, 1996; Beckingham *et al.*, 1996) depending on site characteristics. Roads were assumed to never recover in the modelling in order to conservatively estimate remnant levels within disturbance events. I defined recovered forest as averaging tree height of 3 m; recovery times would obviously be shorter if 1 m or 2 m were used. The definition of when a forest is recovered is entirely dependent on the modelling objectives and in the case of this study, 3 m represented forestry-based standards for tree regeneration (Van Rensen, 2014). These choices influenced the outcomes of some pattern metrics like edge density, remnant levels, and remnant size.

The conservative method that was used to map forest cover changes likely underestimated the observed pattern indices for a number of reasons. The VCT algorithm used to map the forest cover changes has been known to underestimate forest disturbance (Huang *et al.*, 2010). Additionally, a set of criteria were implemented for including forest cover changes in the annual maps, such as persistence of change through multiple years, minimum mapping unit of approximately 1 ha, and large disturbance magnitude. Thus, small, low severity, and ephemeral disturbances were not likely to be considered in the analysis, although these changes also contribute to landscape change. As an example, the estimation of forest edge density was lower than what would be expected given the research of Wulder *et al.* (2008d). These types of discrepancies are likely due to differences in data processing (*i.e.*, VCT and BAP) as well as the objective of assessing large magnitude disturbances. Moreover, the pixel resolution and signal-to-noise ratio for the sensors onboard the Landsat satellites are not well-suited for detecting fine-scale disturbances. Therefore, the analysis is predominantly a stand replacing representation of forest cover change of Canada's boreal forest.

In the future, landscape pattern databases should be developed with higher thematic resolution and with other properties that permit cross-scale comparisons. The classes used in this research were simply *forest* and *non-forest* which were suitable for demonstrating the novel temporal property of the database while maintaining high class accuracy. Increasing the number of thematic land cover classes and, consequently, the number of potential combinations of land cover transitions between classes increases the utility of such landscape pattern databases for applications in many more ecosystems; however, it also significantly increases computational complexity and introduces a greater propensity for map misclassification errors (Langford *et al.*, 2006). Additionally, further research is needed to develop database properties that enable cross-scale comparisons. Scaling has been identified as a key area for research in landscape ecology and specifically the selection of an appropriate scale for a given objective (Wu and Hobbs, 2002). Landscape pattern databases that have multi-scalar properties provide the means to understand the behavior of pattern indices at multiple scales and permit the selection of a scale that is appropriate for any given research objectives (Wu, 2004). The development of future landscape pattern databases should maintain temporal and scaling properties while providing higher thematic resolutions for the broadest possible applications.

7.7 Directions for future research, management recommendations, and new questions

The current state of knowledge of energy development in forests is primarily limited to persistent linear corridors and surface mining. As the number of *in situ* energy projects rapidly increase in forested ecosystems, research programs should focus on the non-discrete impacts of energy development. Moreover, research into the monitoring of pipeline impacts to forested ecosystems will be critical during this period of new expansions.

The forestry industry is often a primary stakeholder in landscapes undergoing rapid energy development and is usually responsible for maintaining forest structure, function, and composition as part of sustainable forest management objectives (Alberta-Pacific Forest Industries, 1999; Schneider *et al.*, 2003). The development of a planning framework that brings all stakeholders together will be a necessary first step towards mitigating the negative impacts on multiple-use landscapes undergoing rapid energy development. The best way to close the gap between the HRV and anthropogenic disturbance patterns is for all natural resource regulators and stakeholders to work collaboratively towards a common, mutually agreed upon set of disturbance pattern goals.

Future remote sensing of energy development should be focused around integrating observations from a range of spectral, spatial, and temporal domains. Relating those data to recurring ground campaigns will provide a sound basis for monitoring, assessing, and establishing reclamation standards that are suited to boreal forested ecosystems. There remain a number of unique opportunities for exploring the impacts of energy development on boreal forested ecosystems. For example, nighttime lights can be used as a proxy for the level of oil and gas activities based on flaring (Elvidge *et al.*, 2009), and disturbance trend algorithms (Hermosilla *et al.*, 2015) can provide new insights into the persistence and recovery success of abandoned mines, well sites, and cut lines (Powers *et al.*, 2015). These data and tools will form the basis for monitoring forest change in landscapes undergoing rapid energy development.

As a stepping stone for future research, I leave the reader with the following questions arising from the investigations in this dissertation that deserve timely answers:

1. Does anthropogenic disturbance alter recovery rates and compositions of boreal forests?
2. Are there landscape structures and thresholds in boreal forests that are irreversible?

3. What are the reinforcing feedbacks that sustain novel ecosystems in boreal forests?

The first question relates specifically to the maintenance of ecological memory and by extension the resiliency of boreal forests. Recovery of boreal forests deserves as much research attention as disturbance if we are to improve our understanding human-dominated forested ecosystems. I stress the priority for this research because forest recovery is a lagged effect that requires decades to produce empirical results (Turner *et al.*, 1998). The second question relates to the persistence of boreal forest spatial patterns. Identifying concrete thresholds in landscape structure that should not be crossed will be key for communicating landscape restoration science to policy makers and land managers (Rockström *et al.*, 2009; Scheffer *et al.*, 2012). The third question relates to the restoration of industrialized boreal forests. There is a large body of literature to support the hypothesis that certain landscape structures are associated with novel ecosystems (Kauffman *et al.*, 2010; Laberee *et al.*, 2014; Lesmerises *et al.*, 2013, 2012; Niemelä, 1997; Van Wilgenburg *et al.*, 2001). Identifying the reinforcing feedbacks that sustain novel ecosystems in boreal forests will be the first step towards restoration of industrialized landscapes.

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