A century of landscape change in the southern Rocky Mountains and Foothills of Alberta: Using historical photography to quantify ecological change

by

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Abstract

Throughout the Rocky Mountain regions of North America fire regimes have been altered towards longer fire return intervals in the 20th century as compared to the 18th and 19th centuries; this has been accompanied by losses of grassland and open canopy woodlands due to encroachment by closed canopy forests. However, we lack information regarding how much grassland or open canopy woodland has been lost, where these losses have occurred, and what has caused them. I examined what we know about fire regimes, and how we incorporate this knowledge into management of forested landscapes. Historical land-based oblique angle photographs taken at the start of the 20th century can provide information regarding what vegetation looked like when humans began to alter fire regimes. Unlike aerial photographs, which are used extensively in remote sensing, the major challenges associated with using historical photographs are that georeferencing methods and image classification procedures are still in their infancy. No one has yet conducted a landscape scale spatial analysis of oblique angle historical photographs to describe vegetation change, and to determine the causes or correlates associated with this change. I developed new methods in GIS and image analysis by using a new software tool (the WSL Monoplotting Tool), and created a new method to extract raster-based data from oblique angle historic photos. I used 137 historical repeat photographs from the Mountain Legacy Project covering 320,000 ha in the southern Alberta Rocky Mountains to measure the vegetation change across the landscape between 1909 and 2008. I found that the majority of the landscape (63.4%) had remained in the same vegetation category, 28% of the landscape was in a later seral stage, and less than 9% was in an earlier state. In 1909 58% of the landscape was non-forest (grasslands, meadows, shrubs, non-vegetated and open canopy woodlands) and 42% in closed canopy forest (conifer, broadleaf deciduous and mixedwood). In 2008 42% was non-forested and 58% forested. The Montane Natural

Subregion had the greatest proportion of area undergoing successional advancement, with substantial (but lesser) forward change in the Subalpine, Alpine and Foothills Parkland. We saw that nearly 37% of historical grasslands and 80% of open canopy woodlands converted to more advanced successional types, and this appears to be due to gradual forest advancement from the historic forest edge. The causes of these changes in vegetation structure were related to topography and disturbance history. I then tested the assumption that the historic vegetation structure at the turn of the 20th century was less susceptible to burning at high intensity and over larger areas than the current vegetation structure. I used the Burn-P3 model to compare burn probability, fire intensity and fire size for two different scenarios: a) the baseline scenario (the landscape as of 2014); and b) a historical restoration scenario (landscape restored its historical (1909) vegetation structure). I used a subset of the photographs I used for examining landscape vegetation change to determine what the historic vegetation composition was in the Bob Creek Wildland. I used indicator kriging to create a seamless coverage of historical vegetation structure. I found that the overall mean burn probability was only reduced by 1.2% in the historical restoration scenario. However, many areas of the landscape showed increased burn probability and others showed decreases in burn probability, and these were largely associated with expected changes in rates of spread associated with different vegetation changes between the two scenarios. Increased burn probability was associated with areas that had changed from forest to grassland, and areas with decreased burn probability were associated with portions of the landscape changed to broadleaf deciduous vegetation. When I only considered areas that would burn at an intensity greater than 4,000 kW/m, I found the historical restoration scenario reduced the mean burn probability of the landscape by 44%. The mean fire size was also reduced in the historical restoration scenario. Many parts of the wildland had burn probabilities that were less than 10% what they were in the baseline

scenario. If managers were to use the historic state of the Wildland as a restoration target, they would see a net benefit with regard to losses due to intense wildfire.

Preface

This thesis is the original work of Christopher Alec Stockdale.

Chapter 2 of this thesis has been published as Stockdale, C., Flannigan, M., and Macdonald, E. 2016. Is the END (emulation of natural disturbance) a new beginning? A critical analysis of the use of fire regimes as the basis of forest ecosystem management with examples from the Canadian western Cordillera. Environmental Reviews 24(3):233-243.

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Chapter 4 of this thesis is in preparation to be submitted as a manuscript with the authors Christopher Stockdale, Ellen Macdonald, and Eric Higgs.

Chapter 5 of this thesis was a collaborative effort between Christopher Alec Stockdale (CAS) and the Government of Alberta Forest Protection Branch's Landscape Wildfire Specialist Neal McLoughlin. CAS was responsible for the concept, study design, image analysis, interpolation, fuel class harmonization, and interpretation of final results, all writing of the chapter, preparation of tables (except for Table 5.3), and preparation of figures 5.1, 5.3, 5.7 and 5.10. Figures pertaining to Burn-P3 inputs (5.2, 5.4, 5.5, 5.6) were all the work of Neal McLoughlin as part of a larger analysis looking at the burn probability of the entire Calgary Forest Management Area. Neal McLoughlin and CAS prepared the output figures together from the model runs (Figure 5.8 and 5.9). The Burn-P3 model was run by Neal McLoughlin with significant input from CAS regarding parameters. Opinions in the paper are those of CAS, and may not reflect the views of the Government of Alberta. When submitted for publication, this paper will be co-authored by CAS (lead author) and Neal McLoughlin (secondary author) with the inclusion of Mike Flannigan, and Ellen Macdonald.

Dedication

Five thousand, seven hundred and seventy three days ago, on November 29, 2000, I handed in my Masters of Science thesis to the library at Oregon State University. The dedication in that thesis ended by saying:

"To Kelly Ness, and her boy Jamie, the light and inspiration for all I do. My new family, here I come…"

I moved to Victoria two weeks later on December 13 and we have been married for nearly 15 years now. Jamie is fully-grown and living on his own. Since then we have added two new boys in the family, Quinn and Owen. Over these years, there have been many challenges, illnesses, moves, career changes, some incredible highs, some painful lows, but one thing that has never changed is the support and love from my wonderful life partner.

This dissertation is only possible because of her support, her love, and her dedication to keeping us whole.

Thank you Kelly, always.

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My committee has provided me with a great deal of support, encouragement and motivation. I thank Ellen Macdonald, my supervisor, for both her patience and her ability to keep the pressure on to do the best work possible. Her understanding through many of the personal challenges that have occurred over these past five years is greatly appreciated, and I thank her for her strong editing skills that have helped me get two papers published out of this already. Eric Higgs has been a colleague and now collaborator for several years, and I thank him for his kindness during my days before my PhD began, and for the opportunities that he has helped find. Mike Flannigan is also an inspiring scientist to have on a committee. All three of my committee members have such wide interests and skill sets that I am most grateful for what you have managed to impart to me.

Just over a year after I started, I was at a remote sensing conference in Oregon, and was still struggling to get the image referencing system I was using to work. I spoke with many image analysts there who more or less said, “Wow, what you want to do sounds cool, I don’t have any answers for you though, that is too hard”. On the final day, I met Werner Muecke from the Technical University of Vienna, and he said, “Oh, have you heard of monoplotting? That will do what you want”. I had not heard that word before, and still have yet to meet another person who knows it. So Werner, thank-you for saying that magic word. Without it I would have never met Claudio Bozzini, and who knows what direction this dissertation would have taken.

Claudio Bozzini has been a massive help through this thesis research. From the moment I first contacted him he was immediately willing to help me learn to use his software. I was his guinea pig and crashed the program repeatedly, which led to many new versions. He took all my requests and wrote them into the base code, and in the end I had a tailor made piece of software just for my project. We had many conversations over Skype and Teamviewer, with me trying to understand his thick Swiss Italian accent, and him constantly asking me to speak slower (I talk fast when I get excited). We got it all to work in the end.

Numerous helpers have been involved in some of the more mundane GIS tasks associated with this project. Margaret Raimann spent a summer building databases, renaming files and filling out tables. Caitlin Mader carried on with this task and did some great work. Cat Boyes was my super-georeferencer-globetrotter-extraordinaire. Without Cat, I would probably have had far fewer images to work with.

I have to thank my parents and sister for all their encouragement. It took me a while to get around to the PhD, but we can now add another Dr. to the family.

Neal Mcloughlin was a powerhouse running the Burn-P3 model for me.

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- A Graduate Research Assistantship, research funding and conference travel from a Discovery Grant from the Natural Sciences and Engineering Research Council (Canada) to Ellen Macdonald
- Alberta Agriculture and Forestry, Wildfire Prevention section for funding the final year of my dissertation

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1.1. Understanding ecological history to aid landscape and forest management

Land managers in Alberta in the 21st century face a wide range of challenges: preservation of ecological functioning, maintenance of biodiversity, providing an economically viable flow of timber and other forest products, managing disturbances associated with mineral and oil and gas activity, ensuring recreational access, and dealing with many other demands on our natural resources and landscapes. Managing these conflicting needs is no simple task. Over the past several decades throughout North America, natural resource management practices have moved from single-issue approaches (sustained yield forestry, natural reserves, agricultural zoning, among others) towards more holistic and system-wide multiple use approaches. With regard to the management of forest landscapes we have seen significant growth in the acceptance of ecosystem based management and the emulation of natural disturbances (END).

Effective implementation of the END model requires a sound understanding of ecosystem dynamics, disturbance regimes, and how these interact to affect ecological change through time. The END model assumes that forest ecosystems are too complex for us to be able to manage them by focussing individually on distinct species and components. Instead, END proposes that by emulating the frequency, size, intensity, and shape of the natural disturbances we can maintain or restore function, structure, and spatial and temporal patterns at the stand and landscape scale, and in so doing, preserve a variety of habitats and biodiversity (Franklin and Forman 1987; Kaufmann et al. 1994; Armstrong et al. 2003). Disturbance regimes and their effects on vegetation structure are highly variable in both space and time, however, and we often do not have as detailed an understanding of historical disturbance regimes as we would like in order to design an effective END system.

Some argue that the past is not a useful ecological model for managing ecosystems into future (Klenk et al. 2008) because anthropogenic climate change will likely bring novel conditions. However, this view assumes that ecological history studies are useful only as blueprints for ecological restoration. Some have argued that the best reason to study the past is that it helps us understand how our present landscape emerged from our previous actions, which in turn will help us predict future outcomes of current actions (Swetnam et al.

1999; Higgs et al. 2014). One of the major challenges with studying ecological history in this context is choosing an acceptable reference point to use as a baseline for comparison to current ecological conditions. There is no single “right” reference point: however, the pre-European settlement period is widely used throughout North America for ecological restoration targets in forest management (Brown et al. 2004; Baker et al. 2007; Barrett et al. 2010; Churchill et al. 2013), rangeland management (Fuhlendorf and Engle 2001) and protected areas management (White et al. 2003; Mawdsley et al. 2009; Bjorkman and Velland 2010; Higgs et al. 2014).

1.2. Changes in disturbance regimes and vegetation structure

In the past century, there has been considerable change in disturbance regimes and vegetation structure throughout western North America (Arno 1980; Barrett 1996; Bradley and Wallace 1996; Heyerdahl et al. 2001; Wright and Agee 2004; Van Wagner et al. 2006; Romme et al. 2009). Forested areas in some foothills and mountain systems throughout western North America are more extensive, dense, and homogenous in structure and composition than they were 100 years ago (Strong 1977; Gruell 1983; Campbell et al. 1994; Brown et al. 1999; Rhemtulla et al. 2002; Fulé et al. 2002; Hessburg et al. 2005; Higgs et al. 2009). These changes could be due to: climatic fluctuation (Johnson and Fryer 1987; Johnson and Larsen 1991; Brown 2006; Daniels et al. 2011; Gedalof 2011); modern fire suppression (Arno and Gruell 1983; Arno et al. 2000; Gallant et al. 2003); or changing land use practices stemming from European settlement changing the relationship between First Nations peoples and their landscape (e.g. bison hunting, prescribed fire use) (Arno and Gruell 1983; Kay 1994; Campbell et al. 1994).

Arno (1980) reviewed the literature pertaining to the Rocky Mountains in Canada and the northern USA and found that pre-1900 fire return intervals ranged between 15-30 years in the Montane Natural Subregion (Natural Regions Committee 2006), and were more widely ranging from 30-150 years in the Subalpine Natural Subregion. Throughout the Rocky Mountains in Canada and the northern USA, very little area has burned since 1910, which was a very large fire year throughout the intermountain west (Pyne and McLean 2008). Heyerdahl et al. (2008) examined fire frequency across numerous sites in the northern Rocky Mountains of the USA, and found frequent fires throughout the region from 1600-1880, after which time fire occurrence began to decline rapidly, with virtually no fire throughout the region from 1920 to the present day. Hawkes (1979), Rogeau (1999; 2004;

2005a; 2005b; 2008; 2016), Tande (1979), and Barrett (1996) have all conducted field studies in the Canadian Rocky Mountains from the USA-Canada border to Jasper National Park and found significant lengthening of fire return intervals and fire cycles between the 1700-1800s and the 1900s, often by orders of magnitude.

Not only have the fire regimes changed throughout the region, so too has the vegetation structure across the landscape. Arno and Gruell (1983) described a large-scale conversion of grasslands to forests between 1936-1981 in southwestern Montana. Gruell (1983) also observed a great deal of forest encroachment on former grasslands and substantial increase in canopy closure within the forested areas between 1871 and 1982 in Montana and Idaho. Rhemtulla et al. (2002) found that grassland cover in the Montane valleys of Jasper National Park had declined by 50% over roughly the same time period (1915-1998). Manier and Laven (2002) in Colorado, and Roush et al. (2007) in Glacier National Park in Montana, also made the observation that forest encroachment into grasslands and meadows was readily apparent over the same time period. Kubian (2013) made similar observations in Kootenay National Park. All of these aforementioned studies used historical repeat photography to show landscape vegetation change.

Other studies also confirm observations of forest encroachment into grasslands. Strong (1977) used pollen from sediment cores extracted from lakes throughout southern Alberta to compare pre- and post-settlement vegetation, and found that what was considered “aspen parkland proper” in the 1970s used to be “groveland” and what was “groveland” in 1977 had previously been fescue grasslands. Campbell et al. (1994) also used pollen to show a biome-wide replacement of grasslands by aspen that began in the 1880s or 1890s along the margin of what is now referred to as the aspen parkland all the way from Regina, through to Edmonton and south to Calgary.

In summary, along the Eastern front of the Rocky Mountains there has been a significant and steady loss of grasslands and open canopy woodlands from the turn of the 20th century to today (Arno and Gruell 1983; Rhemtulla et al. 2002; Heyerdahl et al. 2006). Considering that grasslands and Montane areas are some of the most threatened ecosystems in western North America (Archer 1994; Noss 2013), it is important to understand how much they have changed in order to effectively manage the ecological integrity of these landscapes and meet the needs of timber management, protected areas, multiple use areas, and to mitigate wildfire risk. The knowledge we could potentially gain regarding disturbance regime shifts and vegetation change could be useful to set ecosystem management targets and improve our ability to manage under the END model in general.

Using historical conditions as a restoration target might not make ecological sense in all areas, but not all historic ecosystems are ill adapted to future climate conditions (Jackson and Hobbs 2009).

Disturbance regimes are typically studied using: dendroecological techniques to document fire history (Barrett and Arno, 1988; Marcoux et al. 2015; Chavardes and Daniels, 2016); paleoecological techniques such as analysis of lake sediment cores to examine relative and absolute charcoal abundance over time (Campbell and Campbell 2000; Carcaillet et al. 2001); examination of historical records from fire occurrence databases (Bergeron et al. 2001; Tymstra et al. 2005; Parisien et al. 2006); or are modeled using inputs derived from these other sources (Li, 2000; Wimberly and Kennedy, 2008; Rogeau 2016). Vegetation change is typically studied using many of the same techniques. Dendroecology can be used to describe stand dynamics (Ehle and Baker 2003; Axelson et al. 2009), paleoecological methods can examine pollen (in addition to charcoal) to describe past vegetation composition (MacDonald et al. 1991; Lorenz 2009; Prichard et al. 2009), historical records such as maps (Johnson and Fryer 1987) and aerial imagery can be used to document changes in vegetation (Chuvieco 1999; Rutherford et al. 2008; Fichera 2012; Pickell et al. 2013), and models are widely used to examine vegetation composition and disturbance regime relationships (Keane et al. 2004; Calkin et al. 2005; Wimberly and Kennedy 2008).

Remote sensing can be used to measure changes in patterns, but is limited to the recent past. In most of North America the earliest aerial photography dates to the 1930s, and in the region of western Canada that was the focus of this research, the first systematic aerial survey was conducted in 1949. To look further back in time, some researchers have used historical land based oblique angle photographs (as old as 1870s) to describe ecosystem change. In addition to studies already cited (Arno and Gruell 1983; Gruell 1983; Manier and Laven 2002; Roush et al. 2007; Kubian 2013), Hastings and Turner (1965), Gruell (1980), and Webb (1996) have also used historical repeat photography to describe ecological changes over periods of 80-100 years. The primary drawback of these studies is their observations were not spatially quantified.

1.3. Historical repeat photography and the Mountain Legacy Project

Historical repeat photographs (photographs taken of the same area from the same location at different points in time) have been used for decades to show ecological change

through time, but their use has been primarily limited to descriptive studies. Some of the studies predated the development of usable Geographic Information Systems (GIS) software which limited their ability to conduct spatial analyses. Most of these studies were limited to purely descriptive measures showing changes in vegetation over time. Attempts at quantitative assessments of historical photographs by labor intensive manual procedures have been limited to the small spatial scale of only a few meadows (Roush et al. 2007) or single valleys (Rhemtulla et al. 2002; Watt-Gremm 2007). While Rhemtulla was able to make relative change measurements by using overlays of historic and present day images, she was unable to quantify the absolute area that had changed. Watt-Gremm (2007) pioneered a method to make oblique repeat photography change measurements spatial by overlaying a spatially referenced grid on oblique photographs; however, his observations were limited to seven image pairs covering less than 2,000 ha in total, and due to technological constraints he could not determine the accuracy of his methods.

No one has yet used historic repeat land based oblique angle photographs to conduct a landscape-level (100's-1000's km^2) quantitative assessment of vegetation change across a century. The primary reason for this is due to the inherent difficulty associated with spatially referencing oblique angle photographs as compared with aerial photographs. Early attempts to create computer based methods for spatially referencing such images (Aumann and Eder 1996; Aschenwald et al. 2001; Mitishita et al. 2004; Corripio 2004; Fluehler et al. 2005) have not been widely adopted because they are task-specific, relatively inaccurate, and restricted by available computing power. Corripio (2004) developed a computer program that georeferenced oblique images by allowing the user to manually rotate a DEM to a perspective matching an oblique image. Stockdale et al. (2010) assessed this tool and found that, while cumbersome, it showed considerable promise for evaluating ecological change using historical oblique angle photographs. With a growing number of photograph collections that can be used to measure historical landscape change, developing accurate methods to analyze these images within a GIS would be of great benefit.

The Mountain Legacy Project (Higgs et al. 2009) is a repeat photography project larger than any other similar project in the world with more than 120,000 historical images taken in the mountainous regions of western Canadian by numerous surveyors dating from the late nineteenth to the early twentieth centuries. These images were originally taken in order to develop topographic maps. Survey crews climbed peaks, ridges and promontories throughout the Canadian Rocky Mountains and usually took a series of photographs from each station to create a panorama of what was visible from that location. All photostations

were themselves visible from numerous other photostations. They used large format cameras with 4” x 6” glass plate negatives, which were capable of capturing incredible detail. Within the original photos, it is possible to identify tree species within several kilometres of the camera location, tree densities can be quantified, and the understory vegetation is clearly visible. At greater distances from the camera, coarse scale vegetation patterns are clear in most images. To date, more than 6,000 of these images have been repeated (in the southern Rockies from 2005-present day) from the exact original locations (and are referred to as “paired images”); these provide us a clear view of the changes on the landscape over this time period. While some researchers have used these images to examine ecological change (Rhemtulla et al. 2002; Watt-Gremm 2007; Kubian 2013), their real potential to evaluate large spatial scale ecological change throughout the Alberta Rocky Mountain region has remained largely untapped.

1.4. Goals and objectives of this dissertation

This dissertation focuses on quantifying forest invasion of grasslands and the losses of open canopy forests since the time of European settlement in the late 1800s and early 1900s using historical repeat land based oblique angle photographs. I also examined how these vegetation changes have changed the probability of burning and intensity of wildfires that may occur on this landscape and how we can use knowledge of these ecological changes to manage the landscape. While the study area is focused on the landscape of the southern Alberta Rocky Mountains, given what has been observed by other investigators with regard to fire regime changes and forest encroachment on grasslands and the losses of open canopy woodlands throughout western North America, the methods and findings of this research have broader applicability.

To achieve this goal, I conducted a detailed literature review of how much we know about fire regimes in general and how we could use this knowledge to inform approaches to Natural Disturbance-based ecosystem management (Chapter 2); developed new methods to spatially reference oblique angle photographs (Chapter 3); measured ecological change across a broad landscape using the MLP photos and described potential mechanisms responsible for this change (Chapter 4); and finally, used this information to consider how restoration of the Bob Creek Wildland to its historical condition might change burn probability, wildfire intensity, and fire size distribution compared to the modern day baseline landscape (Chapter 5). These chapters are further described below.

Chapter 2 -- *Is the END (Emulation of Natural Disturbance) a new beginning? A critical analysis of the use of fire regimes as the basis of forest ecosystem management with examples from the Canadian western Cordillera*. In this chapter, I reviewed the literature related to fire regimes, and the END management model. I described what we know about fire regimes in western North America, and what drives their variability. Fire regimes are often used as "coarse filters" to manage biodiversity but this only makes sense if we understand them. I examined how well the END model uses what we know about fire regimes, identified several gaps in knowledge and misapplications of the knowledge we do have, and made recommendations for how to fill these gaps and apply this knowledge more effectively.

Chapter 3 -- *Extracting ecological information from oblique angle terrestrial landscape photographs: performance evaluation of the WSL Monoplotting Tool*. This chapter presents results of testing of the WSL Monoplotting Tool, for its spatial accuracy and its potential use for georeferencing the MLP images, and developed new procedures in GIS for using a raster based approach to classifying vegetation in historical oblique image pairs. The WSL Monoplotting Tool was not available at the start of my PhD studies in 2011, and I worked with the creator of this software, Claudio Bozzini, to modify and tailor this software tool to meet the needs of this research. This procedure became the basis for analyses of images used in the next two chapters.

Chapter 4 -- *A century of landscape change in the southern Rocky Mountains and Foothills of Alberta* . I used the methods developed in Chapter 3 to georeference and classify the vegetation in 137 image pairs from the MLP collection covering a landscape of 320,000 ha. I classified the vegetation into seven discrete categories (conifer forest, broadleaf deciduous forest, mixedwood forest, open canopy woodland, shrubland, grassland, and nonvegetated), measured the changes in these vegetation categories from 1909-2008, and examined correlates of changes to gain insight into what potential mechanisms might be related to this change (elevation, solar radiation, disturbance history).

Chapter 5 -- *Using historic landscape vegetation structure for ecological restoration: effects on burn probability in the Bob Creek Wildland, Alberta, Canada* . I used the MLP photos covering the Bob Creek Wildland, and using interpolation techniques filled in the portions of the landscape that could not be seen in the photographs. I compared the historical vegetation structure to the present day vegetation structure, and I used the historical condition for a hypothetical landscape “restoration” and tested the effect of this restoration on burn probability, fire intensity, and fire size distribution in the Wildland Park. I

used the Burn-P3 model for this analysis, and we implemented new procedures in the model that had not been used previously.

In the concluding chapter, I present an overview of the results of this research and provide recommendations for future research, refinements of methods, and implications for land managers.

Chapter 2: Is the END (Emulation of Natural Disturbance) a new beginning? A critical analysis of the use of fire regimes as the basis of forest ecosystem management with examples from the Canadian western Cordillera.

2.1. Abstract

As our view of disturbances such as wildfire has shifted from prevention to recognizing their ecological necessity, so too forest management has evolved from timber focused even-aged management to more holistic paradigms like ecosystem based management. Emulation of Natural Disturbance (END) is a variant of ecosystem management that recognizes the importance of disturbance for maintaining ecological integrity. For END to be a successful model for forest management we need to describe disturbance regimes and implement management actions that emulate them, in turn achieving our objectives for forest structure and function. We review the different components of fire regimes (cause, frequency, extent, timing, and magnitude), we describe low, mixed, and high severity fire regimes, and we discuss key issues related to describing these regimes. When characterizing fire regimes, different methods and spatial and temporal extents result in wide variation of estimates for different fire regime components. Comparing studies is difficult as few measure the same components; some methods are based on the assumption of a high severity fire regime and are not suited to detecting mixed or low severity regimes, which are critical to END management, as this would affect retention in harvested areas. We outline some difficulties with using fire regimes as coarse filters for forest management, including a) not fully understanding the interactions between fire and other disturbance agents, b) assuming that fire is strictly an exogenous disturbance agent that exerts top-down control of forest structure while ignoring numerous endogenous and bottom-up feedbacks on fire effects, and c) assuming by only replicating natural disturbance patterns we preserve ecological processes and vital ecosystem components. Even with a good understanding of a fire regime, we would still be challenged with choosing the temporal and spatial scope for the disturbance regime we are trying to emulate. We cannot yet define forest conditions that will arise from variations in disturbance regime; this then limits our ability to implement management actions that will achieve those conditions. We end by highlighting some important knowledge gaps about fire regimes and how the END model could be strengthened to achieve a more sustainable form of forest management.

2.2. Introduction

Land managers in the 21st century are faced with a wide range of challenges: preserving ecological functions, maintaining biodiversity, providing economically viable timber flow, managing non-timber forest products, managing exploration for and extraction of mineral and energy resources, and ensuring recreational access. Managing these frequently conflicting demands is no simple task. In response to these pressures, forest management practices have gradually moved away from the one-dimensional "timber-above-all-else" sustained yield model of the "normal forest" that was common in the 1960s (Osmaston 1968), in which there was an equal portion of the landscape in all age classes up to the rotation age of the management area (i.e. if the rotation age were 100 years, there would be10% of the landscape in each 10-year age class to provide an even flow of timber at harvest age). Managers are now moving towards more holistic paradigms such as ecosystem management, one approach to which is the emulation of natural disturbances (END) (Hunter 1993; Sedjo et al. 1998; Spence and Volney 1999; Long 2009; Gauthier et al. 2009; Kuuluvainen and Grenfell 2012). While this change is welcome from an ecological perspective, the success of END hinges upon how well management actions mimic natural disturbances, which in turn depends upon the level of understanding of how disturbances regulate ecosystems.

This paper is a critical examination of both the END concept and how well we apply fire regime characteristics to this form of management. We clarify the components of fire regimes, describe what we do and do not know about them, identify key knowledge gaps, and address some common misconceptions regarding how fire regimes regulate ecosystems. We also discuss the risks of applying the END model in the absence of local information on fire regimes over a long enough time period to avoid issues associated with the fire suppression era beginning in the early to mid-1900s, and understanding the severity of the fire regime and its effect on ecological processes. We illustrate this with examples from the Canadian southern Rocky Mountains region, where wildfire is the predominant natural disturbance agent (McCullough et al. 1998; Stocks et al. 2002; Weidinmyer and Neff, 2007; Government of Canada 2015*a*, 2015*b*), although the concepts explored herein apply well to cordilleran systems in general, and to varying degrees to other forest regions in which wildfire is the dominant natural disturbance.

2.3. Management vs Science

Over the past century, approaches to managing forests and other natural resources have been influenced both by our scientific understanding of forests and by forest management paradigms. Advancements in scientific knowledge are generally followed by changes in management practices while, in turn, forest management concerns are an important driver of research agendas.

Historically, concepts of forest ecosystem dynamics were relatively simplistic and linear and disturbances were viewed as something undesirable. Whether these disturbances were fires, outbreaks of insects or disease, windthrow events, or some other ecological agent causing damage or death, they were seen as interruptions of the "natural" trajectory of ecosystems to move towards a desirable stable “climax state”, which was the inevitable end-point of forest succession (Clements 1936). Disturbances were seen as events that caused deviation away from the norm in ecological systems (Rykiel 1985), and therefore they were something to be prevented or controlled. Managers held similar views about the role of disturbance in ecosystems, their perspectives being reinforced by the state of scientific knowledge. Forest management efforts were (and arguably still are) primarily focused on maximizing economic returns. Forest stands have maximum economic value at some point along their growth trajectory, often well before the ecological climax is reached. To managers, the climax forest was considered "decadent" and required renewal to ensure a steady flow of economically valuable timber (Gadow et al. 2000).

In recent decades, our concepts of the role of disturbance in ecosystems and the complexity inherent in ecological succession have changed dramatically. Following Clements’ (1936) view of a monoclimax, Tansley (1939) put forth the polyclimax theory which asserted that external forces (such as disturbances) have a significant influence on vegetation successional pathways and can prevent a site from ever reaching its “climax”.

Odum (1969) elaborated upon the idea of a polyclimax, and proposed that succession should be viewed as a complex set of interacting processes, some of which oppose each other. He (Odum, 1969) discussed the concept of "pulse stability", whereby an ecosystem may be viewed as stable in the face of constant/regular perturbation, and this view aligns with Holling (1973) who described that a stable ecosystem can exist in multiple states, and it is the spatial diversity in the system provided by the landscape mosaic that makes it resilient. Successional theories developed further over the past few decades (these are well reviewed in Kimmins, 2004), and at the same time ecologists began to recognize

disturbance as an inherent and natural property of ecosystems (White 1979; White and Jensch 2004). Key aspects of this shift in thinking include the recognition that the ‘climax’ may never be reached due to the disturbance regime interacting with a mosaic of habitats and that a "stable" system could be one that oscillates between two or more states on a regular basis, or in response to a predictable or common disturbance. This is illustrated in forest landscapes in which wildfire regularly resets the successional stage to some condition depending on the severity of the fire; these landscapes are thus comprised of a mosaic of stands with different age class distributions and different dominant tree species depending on the time since and the severity of the most recent fire or other disturbance.

Indeed, this can be seen in landscapes that include both forest and grassland cover and in which new grassland areas are being created by disturbance in one location, while grasslands are succeeding towards forest cover in other locations. Another important change is the scale at which we view disturbance and succession; the dynamics of succession and the effects of disturbance on a single site/patch/community are fundamentally different than what we see at a larger scale, where the importance of disturbance becomes much more clear (White, 1979).

In 1910, US Forest Service Chief Henry Graves stated that the most important element of forestry was to prevent forest fires (Stephens and Ruth 2005). This view of eradicating disturbance seems almost foreign now; most managers accept that disturbance is integral to ecosystem structure and function, and in some cases is an intrinsic property of the system itself (White 1979; Rykiel 1985). As understanding of disturbance and ecosystem dynamics has evolved, so too have management practices. The principles of ecosystem management have developed over the past three decades (Franklin and Forman 1987; Grumbine 1994; Jensen and Bourgeron 1994; Christensen et al. 1996). While ecosystem management approaches are understood and implemented in different ways (Lackey 1998; Butler and Koontz 2005; Bormann et al. 2007; Gustaffson et al. 2012), they all broadly reference the triple bottom line definition of sustainability in which management of natural resources must be ecologically, socially, and economically sustainable (Kaufmann et al. 1994). One approach to implementing ecosystem management, which has seen dramatically increasing popularity in the past two decades, is to emulate natural disturbances (the END model) (Hunter 1993; Bergeron et al. 1999; Spence and Volney 1999; North and Keeton 2006; Long 2009, Gauthier et al. 2009).

2.4. Defining END

2.4.1. *Emulating Natural Disturbance*

The END paradigm of forest management arose in different regions of North America as an approach that recognized the complexity of forest ecosystems and the inherent role of disturbance as a driver of their ecology (Franklin and Forman 1987; Hunter et al. 1988). The END approach is based on the assumption that by emulating the attributes of natural disturbances, it is possible to maintain or restore ecological structure and function in managed forests and thus maintain biodiversity along with other ecological goods and services (Kaufmann et al. 1994; Armstrong et al. 2003; North and Keeton 2006). The END model can be related to the coarse filter approach to conservation (Noss 1987; Armstrong et al. 2003**),** which initially proposed that if a representative sample of all communities is preserved, that 85%-90% of individual species would be maintained on the landscape. For species that were not "caught" in the coarse filter, a complementary fine filter approach could be used to specifically manage for their needs. Noss (1987) recommended that the coarse filter concept would be more effective by focusing on larger spatial scales than communities (ecosystems, landscapes), and also recommended incorporating disturbance regimes as management elements. This shifted the focus from managing the elements on the landscape (communities, species) to managing both the elements and the processes that created/regulated them. END assumes that ecosystems that are regulated by fire and other natural disturbances are populated by species adapted to these disturbances.

Effective implementation of END requires that the historical disturbance regime be well enough known that we can develop and implement management actions that mimic that regime. With detailed knowledge of natural disturbance regimes managers can design forest management practices so that total volume harvested, harvest block size distribution, rotation lengths and time between stand re-entry, levels of retention within harvest units, and post-harvest site treatments are designed to more closely resemble the patterns created by natural disturbance. One of the inherent weaknesses of the END model is that the description of the historical disturbance regime is limited by the available information, which is often not as comprehensive as necessary.

2.4.2. ***Understanding Disturbance: Fire Regimes***

In much of North America, and in mountain regions particularly (including Alberta, British Columbia, Montana, Idaho, Wyoming, Colorado, and Utah, among others) wildfire is the dominant high-mortality disturbance agent of forested (Habeck and Mutch 1973) and grassland ecosystems (Rowe 1969). While insects may affect a larger area on an annual basis, especially with the ongoing mountain pine beetle, spruce beetle, and spruce budworm outbreaks (Candau and Fleming 2005; Safranyik and Wilson 2006; Bentz et al. 2010), their overall effect through time with regard to restructuring ecosystems is lower than fire (Government of Canada, 2015*a*, 2015*b*)**.**

A fire regime is the interaction of fire with the environment in space and time (Heinselman 1973; Morgan et al. 2001). As described by Moritz et al. (2011), the fire regime is influenced by vegetation, ignitions, and climate. While there is some disagreement on the precise attributes of a fire regime, and even greater disagreement on how to measure them, there is general consensus that the important components are: 1) cause, 2) frequency, 3) timing (seasonality), 4) extent (size), and 5) magnitude (intensity/severity). These can be further sub-divided; e.g., intensity and severity are often discussed separately. Herein, however, we focus our discussion around these five primary attributes, all of which interact with one another, often in a complex fashion.

2.4.2.1. Fire Regime Attributes

Cause: Cause can be considered the highest level, or first order, attribute of fire regimes, as this determines when, where and how often fires burn on the landscape. Fires can be ignited by lightning or by people. Lightning fires are driven by fundamentally different parameters than anthropogenic fires; they burn at different frequencies, in different places, and start at different times of year. Humans have a long history of fire use throughout the world (Marlon et al. 2008; Bowman et al. 2011; Coughlan and Petty 2012). In the foothills regions and grasslands adjacent to the Rocky Mountains, human fire use has been common for millennia. In the boreal, on the other hand, fire use by first nations was practiced (Lewis 1978) but it was perhaps less common, and its influence is debated (Clark and Royall 1995; Campbell and McAndrews 1995). It is very difficult to quantify the proportion of historical fires that can be attributed to First Nations versus lightning.

Frequency: Fire frequency is a measure of the rate at which fires occur on the landscape. Frequency *sensu stricto* is the number of ignitions (the start of a fire of any size) per unit

time per unit area. Fire frequency measures reported in the literature often incorporate elements of area burned as well. A wide variety of metrics are used to describe frequency, such as the hazard of burning (probability of a given location burning per unit time), burn rate (amount of the landscape burned per unit time), fire cycle (length of time required to burn an area equivalent to 100% of the study area), and mean fire return interval (average amount of time between fire events occurring in a given area) (see Romme, 1980 for a thorough description of these various metrics). Fire frequency is one of the most studied attributes of fire regimes, and managers need to pay attention not just to the mean frequency, but its variability in both space and time.

Timing: The time of year at which a fire occurs affects the fuel and weather components of fire behavior, which in turn affect the extent and magnitude of the disturbance. The length of the fire season also influences the number of fires that can occur each year. Timing has an important influence on post-disturbance succession, which in turn may affect future fire behavior.

Extent: Understanding the extent (or size) of fires is important to understand the area affected, and the distribution of patch sizes on the landscape. Furthermore, fire sizes are incorporated into many measures of fire frequency. Many fire regime studies have examined the fire size distribution (Van Wagner 1978; Cumming 2001; Barclay et al. 2006; Moritz et al. 2011), which is required to determine fire cycle, hazard of burning, or fire return intervals. An accurate model of the distribution of fire sizes is important as many older fires have been partially or fully erased by newer fires (data are censored).

Magnitude: Magnitude is a measure of the amplitude of the disturbance. Fire intensity and fire severity are often used interchangeably to describe the magnitude of the fire regime, but these are not the same thing. Fire intensity is a measure of the radiative force of a fire as it burns, and is usually measured in units of energy per unit area per unit time (Keeley 2009). Fire severity is more commonly considered to be the ecological effects of the fire, such as the mortality rates or consumption of duff; these, in turn, affect post-disturbance biological legacies, and vegetation composition and pattern (Keeley 2009).

2.4.2.2. Classification of Fire Regimes

Fire regimes can be classified into general categories. The most common classification methods use variations in frequency and severity (Agee 1993; Arno et al. 2000) resulting in three basic categories of fire regimes:

1. low severity/surface fire
2. mixed severity fire
3. high severity/crown fire.

The general rule in forested ecosystems is that low severity fire regimes have higher frequencies, and high severity fire regimes have low frequency.

Low Severity: Low severity fire regimes are found in grasslands (Brown et al. 2005) and savannah/open woodland systems with widely spaced trees, little ladder fuel, and relatively light surface fuel loading (Arno and Gruell 1983; Heyerdahl et al. 2007). Low severity fire regimes are considered "stand maintaining" because they remove competing vegetation and maintain the dominant canopy (Agee 1993). The fire cycles and fire return intervals in low severity regimes tend to be relatively short; fuel loads are quickly eliminated, and cannot build up to sufficient volume to cause the higher severity fires that result in overstory mortality. Fuel burnt in low severity fire regimes is often grass, herbaceous vegetation, saplings, or shrubs. This type of fire regime requires a frequent and somewhat regular ignition source that coincides with flammability of the fuel. Ecosystems experiencing low severity fire regimes often occur in locations with weather patterns characterized by regular lightning storms; however, they can occur in lightning shadows if the ignition source is anthropogenic, in which case they coincide with areas of frequent human use (Tande 1979; Wierzchowski et al. 2002).

Mixed Severity: Mixed severity fire regimes are complex. There are two intergrading types: temporal mixed severity, and spatial mixed severity (Schoennagel et al. 2004, Agee 1993). Temporal mixed severity fire regimes show alternating fire behavior between low and high severity over time. In general, a temporal mixed severity fire regime is caused by variability in climate and/or interactions with other disturbances that result in significant variation in vegetation composition and fuel loading over time (Carcaillet et al. 2001). Spatial mixed severity fire regimes show differential burning severities across the landscape driven primarily by variation in topography and fuels (Schoennagel et al. 2004). To further complicate the understanding of spatial mixed severity fire regimes, some researchers differentiate between mosaic fire regimes and "pure" spatial mixed severity fire regimes. In the former, patches of fully burned forest are intermixed with patches of unburned trees within the fire perimeter, whereas in the latter there is 25%-75% crown mortality evenly distributed within the fire perimeter. The mosaic fire is one with intermittent crown fire activity whereas the spatial mixed severity fire is one with intermittent candling (Barrett et al. 2010).

High Severity: High severity fire regimes are dominated by crown fire activity. They tend to kill the majority of overstory vegetation within the fire perimeter. Regions characterized by high severity fire regimes tend to have lower variability in topography and fuel structure although there are some mountain environments with high severity fire regimes (Parisien et al. 2006). High intensity crown fires maintain the landscape in patches with the dominant overstory trees in even-aged, single-species stands, but this depends on the time since fire; stands may succeed towards a more mixed composition before the next fire occurs. These fire regimes are often referred to as stand-replacing fire regimes (Agee 1993).

These fire regime "types" exist along a continuum, and the full range of these regimes can be found along the eastern slopes of the Rocky Mountains (Arno 1980; Arno and Gruell 1983; Collins 1992; Arno et al. 2000; Margolis et al. 2007; Amoroso et al. 2011).This makes using fire regimes to guide END planning considerably more complex. We are seeing growing evidence that mixed severity fire regimes in foothills and cordilleran forests are more common than we previously believed (Arno et al. 2000; Amoroso et al. 2011; Kubian 2013; Marcoux et al. 2015; Chavardes and Daniels 2016), and this has significant implications for how we would emulate natural disturbance.

2.5. Current Issues with Characterizing Fire Regimes

Fire regimes are a complex expression of the interactions between fire, short term weather, climate, soils, topography, human history, grazing, browsing, and vegetation succession. Fire regimes are highly variable over time and space, and the numerical values of the various parameters are sensitive to the temporal and spatial scale of analysis (Jelinski and Wu 1996; Senici et al. 2010; Whitlock et al. 2010), as well as the methods used to study them. When the spatial boundary of a study area is changed, or the temporal depth is modified, these fire regime statistics change. Falk et al. (2007) suggest that fire regimes can only be understood by conducting research at multiple scales. Furthermore, approaches to characterizing fire regimes vary, and this leads to conflicting information. Differences among studies in values for fire regime attributes can arise due to differences in the spatial or temporal unit of analysis and the methods of calculating the attribute itself.

Fire regimes can be characterized based on (adapted from Tymstra et al. 2005): 1. Field studies (point sampling, stand origin mapping, dendrochronology), 2. Fire records analyses (fire occurrence database analyses) or, 3. Modelling. The spatial scale over which

each of these approaches (field studies, fire records analysis, and modelling) is most useful varies, and an overview is provided by Morgan et al. (2001).

Field studies are limited to the researcher's ability to find evidence of past fires (fire scars on trees, release in tree rings, charcoal in soil or sediments). In the case of fire scars and tree rings, the temporal depth of analysis is restricted to the maximum lifespan of the trees in the area, which varies by species (Burns and Honkala 1990). Dead wood (snags and logs) can extend this time period if material is well preserved, but this is often not the case. Further complicating field sampling is that evidence of older records is often erased by newer fires, as older trees that may contain scars from earlier fires are consumed by newer fires. Fire scars at one location can be used to define point measurements of fire return intervals, or they can be sampled over a larger area to develop stand origin maps. Fire scars can be sampled to define the boundaries of historic fires. This is not problematic in and of itself, however, fire scar sample locations are often used to define the spatial extent of a given fire/series of fires without regard to survivorship within the fire, thus assuming stand-replacing fire is the dominant fire regime. Of course, this can be overcome with development of detailed age class structure inside the historic fire boundaries to demonstrate the proportion of forest regeneration created by the disturbance events, but this is rarely done. Field studies that use charcoal to date fires have the advantage of being able to look much further back in time (Campbell and McAndrews 1995; Carcaillet et al. 2001), however, they cannot provide evidence with regard to severity, spatial extent, or the landscape pattern of the fires.

Using fire records to determine fire regimes is limited to an even shorter time frame than field studies, as studies are temporally limited to the period during which fire management agencies have been storing such records. In Alberta, for example, the earliest recorded fire statistics are from 1931. Analysis even over this time period is complicated because standards of data acquisition for size limits, how fire boundaries are drawn (which affects statistics related to fire size), fire causes and other attributes have changed over the years; thus fire records from the 1930s do not correspond well with fire records today (Tymstra et al. 2005). Bergeron et al. (2001) combined using archival records with dendroecological data to reconstruct fire frequency in Eastern Ontario and Quebec which helped to extend the temporal window limitations of using the archival data alone.

The use of modelling to characterize fire regimes tends to be focused on using fire growth models to simulate fire spread (and ultimately fire size distributions) (e.g. Li 2000; Rogeau 2005), burn probability (ultimately measuring the rate of annual burning, or the

inverse of the fire cycle) (e.g. Parisien et al 2005), or disturbance models to simulate age class distributions of forest stand structure (e.g. Andison 1998). In all cases, these models must be calibrated by using data from either field studies or fire records analyses, and so they too suffer from the above-mentioned deficiencies in quantification of fire regime parameters.

Differences in methods and spatial scale used to characterize fire regimes nearly always results in varying estimates of the different parameters of a regime. For example, the literature on fire regimes for the eastern slopes of the Rocky Mountains in southern Alberta is replete with discrepancies for nearly all parameters; many researchers have investigated fire frequency and have found different estimates of the fire cycle or fire return interval of the region (Hawkes 1979; Tande 1979; Arno 1980; Johnson and Fryer 1987; Barrett et al. 1991; Andison 1998; Rogeau 1999). These values sometimes vary by orders of magnitude. For example, the modern fire cycle (1961-present) for the Upper Foothills Natural Subregion of Alberta was calculated by Tymstra et al. (2005) to be 627 years and by Rogeau (2010) as 51,772 years. Both studies used the same fire occurrence database, but the spatial boundary of the study area differed between the two. Tymstra examined fires occurring in the entire Upper Foothills Natural Subregion (~21,000 km^2), and Rogeau examined only the portion of the Natural Subregion (~580 km^2) that was contained within a particular forest management unit that intersected the Natural Subregion. Due to the spatial variability in fire occurrence across the Upper Foothills Natural Subregion, the results are different by two orders of magnitude but technically neither of these estimates is incorrect.

Developing stand origin or time since fire maps assumes that the fire regime is a high severity one (Barrett and Arno 1988). Sampling methods designed for high severity fire regimes are not only inadequate to quantify frequency in areas with significant mixed severity fire activity; these methods do not allow for the detection of low severity fire regimes at all. This leads us to the point that there is considerable disagreement over the severity of fire regimes in the mountainous regions of western North America. There is growing evidence that there are both spatial and temporal mixed severity fire regimes in the region (Arno et al. 2000; Amoroso et al. 2011; Mori and Lertzman 2011), but the main methods of measuring the fire regimes – using stand origin maps – makes it impossible to detect the mixed severity fire regimes. This not only affects our ability to detect mixed severity fire regimes, but we underestimate fire frequency altogether because we are not sampling adequately to represent areas burned at lower severities (Amoroso et al. 2011; Marcoux et al. 2015).

Much of the variability in estimates of fire regime attributes is due to variation in the spatial and temporal windows of analyses. As the spatial unit of analysis and the temporal depth change, the fire "regime" will also change. This is a phenomenon akin to the Modifiable Areal Unit Problem (MAUP), which is a well-understood problem in geography and spatial analysis (Jeliniski and Wu 1996). For example, one researcher examines fire regimes in the Subalpine at the exclusion of any other Natural Subregion while another examines a larger area that encompasses the same Subalpine region as well as the neighbouring Montane and Upper Foothills. In this case, they are bound to come up with different measures of fire regime within the Subalpine based solely on differences in which data they have collected and analysed. Rather than focusing on which one is right and which one is wrong, it is more appropriate to focus on which one is most applicable to a given problem.

2.6. Misunderstandings about the Effects of Fire Disturbance

The END model loses effectiveness as a management tool if fire regimes are not well understood. Regardless of the quality of the scientific research, land managers are often too busy to stay on top of the scientific literature, or they are not well versed in the nuances of the fire regime literature itself. They frequently make assumptions about how disturbance works, how systems respond to disturbance, and what the effects of emulating disturbances will be. Some of these assumptions are not always valid:

1. We know enough about fire regimes so that harvesting designed to emulate size, frequency, and severity can be used as a "coarse filter" to manage biodiversity.
2. Disturbance is strictly exogenous, and regulates ecosystems from the top down. Biodiversity is the result of top-down, hierarchical processes. Larger elements (landscapes, ecosystems) control smaller elements (communities, populations).
3. Recreation of pattern is a way of preserving the components and processes inherent to the ecosystem.

2.6.1. *Assumption 1: We know enough about fire regimes to use them as coarse filters*

While it is obviously necessary to use fire regime to guide END planning in many forest regions, the whole fire regime needs to be considered, and this is frequently not done. In addition to using extent and frequency as metrics for emulation of natural disturbance

(Carlson and Kurz 2007; Andison 1998; Li 2001; Pickell et al. 2013; Belleau et al. 2007), magnitude (severity) must be incorporated because this impacts the shape of harvest units as well as the levels of in-block retention. In many locations, and in the southern Alberta Rockies in particular, we simply do not understand the severity of disturbance regimes well enough to use them effectively, nor do we understand them over a long enough time period to appreciate the variability in frequency and extent.

While fire is the dominant disturbance agent of the landscape, it is not the only one: there are innumerable feedbacks between fire, grazers, disease, and insects (Veblen et al. 1994; Bachelet et al. 2000; Johnson and Cochrane 2003; Sankey et al. 2006; Dordel et al. 2008). Even if we did understand fire regimes well enough, we overlook these complex feedbacks at our peril. For example, the east slopes of the southern Rocky Mountains of Alberta are on the boundary of the Great Plains. This area is forested but contains large amounts of grasslands. We know very little about historic disturbance regimes of the grasslands, but a variety of sources show grasslands are being lost (Rhemtulla et al. 2002; Higgs et al. 2009; Stockdale et al. 2015) due to lengthening of fire return intervals (Arno and Gruell 1983; Brown et al. 2005; Sankey et al. 2006; Noss 2013). Along the ecotone of the grassland and foothills forests, there were formerly large herds of bison (Brink 2008) and aboriginal people with a long history of fire use (Lewis and Ferguson 1988; Kay 1994; Boyd 2002; Bowman et al. 2011). The interactions between bison and elk grazing, anthropogenic fire use, wildfire, and landscape level vegetation structure are complex (Campbell et al. 1994; Knapp et al. 1999; Gates et al. 2010), and poorly understood. Due to these interactions, management actions designed to mimic the fire regime alone may have unintended consequences.

2.6.2. *Assumption 2: Disturbance is strictly an exogenous process that exerts top-down control*

Fire consumes vegetation and can be viewed as a herbivore (Bond and Keeley 2005). This leads some to see it as exerting top-down control over the system, but this ignores numerous bottom-up processes that influence how fire behaves and how much vegetation it consumes. The behaviour and effects of the fire are influenced by vegetation composition (fuel) resulting from the severity of previous disturbances (not only fire) and post-disturbance succession (Krawchuk and Cumming 2009). Fire is by no means the only force acting to shape vegetation composition and structure through time. Other endogenous disturbance agents (such as insects, grazers, and diseases) interact significantly to alter fire

behaviour by altering the vegetation composition (McCullough et al. 1998; Bachelet et al. 2000; Johnson and Cochrane 2003, Wierzchowski et al. 2002; Dordel et al. 2008). These interactions are not always linear, nor are their effects always predictable (White 1979; Bousquet and Le Page 2004; Messier and Puettmann 2011). Availability of resources (bottom up controls) such as nutrients and water, among others (all of which can indeed be altered by fire (Certini 2005)), also influence vegetation growth. With more vegetation, there is more fuel, and with more fuel, fires burn with greater intensity. There is considerable risk that when trying to emulate the disturbance we will not achieve the desired results if the mechanisms by which disturbance regulates systems are oversimplified. Treating disturbance as strictly an exogenous process presumes a hierarchy of scale in which biodiversity is regulated by the top-down control of fire, when reality is decidedly more complex.

2.6.3. ***Assumption 3: Recreation of pattern can effectively preserve components and processes***

The use of natural disturbance patterns is important in END management, however it is only the coarse filter and must be accompanied by fine filter approaches if the maintenance of biodiversity is a management goal. While many organisms respond to vegetation patterns, natural patterns alone cannot be expected to create the habitats and conditions required to preserve all species and maintain biodiversity across the landscape (Franklin and Forman 1987; Hansen and Urban 1992). Forest community structures are complex, and cannot be expected to be produced by simply replicating the shape and size of natural disturbances (Lindenmayer and Laurance 2012; Webster 2013). Pattern emulation also fails to effectively emulate many important ecological effects of fire disturbance (Certini 2005). Post fire vegetation communities, biological legacies, and nutrient cycling processes are markedly different than those resulting from forest harvesting (White 1979; Turner 1989; Macdonald et al. 2001; Lafortezza et al. 2006).

Plants show a variety of mechanisms by which they are adapted to disturbances such as fire (Rowe 1983). Some elements of fire disturbance can be mimicked by harvesting (exposed soil, removal of surface biomass, changes in light conditions) such that species' responses to the two disturbances could be similar. Other aspects of the disturbance are not duplicated at all, including the heat generated by fire (which can trigger flowering, seed production, seed release, germination, and sprouting) and chemical changes to nutrient profiles. Plants that can resist fire are often removed in harvesting operations. For these

reasons post-harvest regeneration and succession is likely to differ from what would occur post-fire. There is a very real risk of lowering the resilience of ecosystems by ignoring the deficiencies associated with assuming that mimicking pattern will preserve process. Recreating patterns can be considered to be the coarse filter, but we also need to consider the fine filter approaches to deal with the deficiencies created by using pattern alone.

2.7. Challenges in Applying END to Management

If END is viewed as a tool to deliver ecosystem management, we need to be able to understand how fire impacts ecosystems at multiple temporal and spatial scales. Furthermore, we need to be able to choose a desired future outcome that is within the bounds of what would be considered natural for the area in question (i.e., within the Natural Range of Variation or Historical Range of Variation) (Hessburg et al. 1999; Cissel et al. 1999). Finally, we need to know whether our management actions mimic the effects of fire and will achieve the desired outcomes. All of these are challenging.

Firstly, there are challenges with describing the historic disturbance regime. Whether one chooses to use the term Natural Range of Variability (NRV) or Historic(al) Range of Variability (HRV), these approaches propose essentially the same thing as the END model; by managing landscapes within their ecological boundaries and limits, we can move towards more ecologically sustainable management (Franklin and Foreman 1987). Which reference period and location and which spatial and temporal unit of analysis are appropriate to the management problem? As mentioned above – fire regimes are highly variable in time and space and their characterization is highly dependent on the methods employed and the reference period, location and spatial and temporal unit of analysis. Therefore, it is very difficult to provide a benchmark for their emulation.

Secondly, there are challenges with defining desired future forest conditions. Disturbance regimes are a primary driver of the forest condition (age structure, species composition), and under END the forest condition arising from the documented fire regime becomes the target for management. However, given the extensive variability of fire frequency, the lack of knowledge of historical fire causes, timing, and size class distribution, and uncertainty regarding severity, it is challenging to define a desired future forest condition. We don’t always know the relationship between a certain fire regime and its forest condition through time; we don’t know which time period to use as our target for. Even if the gaps in collective knowledge of historical disturbance regimes could be filled in, does this

describe what the future system should look like given climatic change and its potential (and largely unknown) effects on the future range of variability?

A frequent criticism of the END model is based upon the belief that END proposes that historical landscapes should be reconstructed in the present and used as a model for the future (Klenk et al. 2008; Kramkowski 2012). However, this is a fundamental misunderstanding of the END model; using past disturbance patterns and processes as a template for the current and future landscape is only one possible variant of how END can be used. Swetnam et al. (1999), and more recently Higgs et al. (2014), suggest that the past can be used as a guide rather than a template for the future. "Historical perspectives increase our understanding of the dynamic nature of landscapes and provide a frame of reference for assessing modern patterns and processes" (Swetnam et al. 1999, p. 1189).

Finally, there are also significant challenges with implementing the required actions to achieve our end goals. With so much uncertainty surrounding fire regimes it is difficult to determine what a sustainable future landscape condition would be, and without that, it is virtually impossible to choose effective management options. While it is appealing to use END as a model for a more sustainable form of forest management, without clear targets we are bound to miss the mark.

2.8. Challenges in Operationalizing END Within the Context of Fire Regimes

Even if all the knowledge gaps regarding historic fire regimes could be filled, some fire regime attributes are relatively easy to emulate, some fire regime attributes are considerably more difficult to emulate, and some attributes may not be important to emulate at all if the long term consequences are determined to be unimportant. We have the knowledge and tools to mimic some attributes of fire regime while others require more research and trials.

Cause: The different causes of fire on the landscape do not particularly matter to the END model. The primary differences between anthropogenic and lightning caused fire are in how they have affected the location, timing, size, frequency and severity of fire. It is these components that are more critical to emulate; assuming we know them, cause can largely be ignored.

Frequency: *Sensu stricto*, frequency is a very easy attribute to emulate, assuming we know how frequently fire has occurred in the relevant time frame and area of interest. To be useful for management, frequency must be combined with extent. Successful END Management

will require representation of the variability in size and frequency of disturbance events. This can inform rotation lengths, timing between disturbances, and the size of the disturbance units under the END forest management scheme. Managers need to pay attention not only to the mean of the fire frequency and size, but also their variability.

Timing: While the timing of fire (i.e. spring fire regimes versus summer fire regimes) affects fire extent and severity, emulating the timing of fire is largely impractical. While it is clear that timing of disturbance strongly influences post-disturbance successional pathways, restricting forest operations to narrow time windows would not work very well, nor would emulating the timing of fire by harvesting necessarily result in the same post-disturbance vegetation development. If specific post-disturbance vegetation communities that arise from fire cannot be obtained via harvesting, post-treatment broadcast burns at the right time of year or seeding may work to better replicate the effects of fire.

Extent: This is perhaps the easiest element to emulate, as fire sizes (and their variability) can be directly related to harvest unit size. Extent combined with frequency can determine appropriate harvest block size distributions and rotation lengths.

Magnitude: Assuming we know the proportion and locations of the landscape that experience low, mixed, or high severity disturbance we can emulate these through harvesting and silvicultural prescriptions that modify the structure of forest stands. However, this is one of the attributes we know the least about, and is a vital one to improve upon. From the EMEND experiment we know that even small amounts of harvesting disturbance have negative effects on biodiversity, and that varying levels of disturbance have a very real effect (Caners et al. 2013; Craig and Macdonald, 2009; Pinzon et al. 2012). If indeed more of the landscape is driven by mixed severity fire regimes than we previously thought, this greatly affects management decisions regarding retention levels in harvest units. Some effects of varying disturbance magnitudes are more difficult to emulate, as fire has not only physical effects (post disturbance structure), but chemical effects too (Certini 2005).

2.9. Knowledge Gaps

Not all areas have the same gaps, but we have complete fire regime parameter knowledge in very few areas. The following are a list of some of the key knowledge gaps that often need to be considered:

1) **Mixed severity fire regimes**: which proportion and locations of the landscape experience mixed severity fire regimes? We have likely underestimated fire

frequency because we have not detected how much of the landscape burned at lower severities. As previously indicated, growing evidence suggests that mixed severity fire regimes are more common throughout Alberta in the Rocky Mountains (Chavardes and Daniels, 2016), foothills (Andison, 2012; Amoroso et al. 2011) and boreal regions (Andison and McCleary 2014), and may apply over a much larger area than previously believed.

2) **Low (surface) severity fire regimes**: we have virtually no information regarding how much historical surface fire there was in both forested and grassland systems.
3) **Effects of modern fire suppression and exclusion**: we need to consider the effects of both extinguishing wildfires, and excluding the anthropogenic fires that would have occurred historically. These have both changed the fire regime, resulting in a landscape that has higher connectivity of mature forest, less open canopy forest, and fewer meadows and grasslands (Baker 1992; Arno et al. 2000). In turn, this has possibly created conditions where future fires will burn at higher severity and over larger areas than they would have historically (Bergeron et al. 2004, Adams 2013; Arno et al. 2000). This altered regime also makes it hard to determine the historic landscape structure and disturbance regime.
4) **Lack of temporal depth**: as described above most fire regime studies lack the temporal depth necessary to really understand variability of the various regime parameters through time. While paleoecological studies provide long term information regarding variability in the total amount of fire, we still lack information on how variable severity, size, frequency, and timing of fire over periods longer than ~200 years. While fire scar analyses may reveal fires that burned in the early 1800s and into even the 1700s or 1600s in parts of western Canada, the number of samples declines rapidly the further back in time we look.
5) **Lack of spatial coverage**: we have essentially no information on fire regimes for many areas of the landscape, especially regarding the time period before modern records were kept (which is in the suppression era). Field studies are labour and cost intensive, and have only covered a fraction of the total landscape in western Canada. Given the spatial heterogeneity of fire, we cannot assume that the fire regime of one region is similar to that of a neighboring region. Climate patterns, topography, large animal grazing and browsing, and pre-European settlement human activity all play important roles in the fire regime in a given area (Romme

and Knight 1991; Brown et al. 1999; Bachelet et al. 2000; Brown 2006; Boucher et al. 2014). Different configurations of these factors can lead to unique outcomes.

2.10. Moving Forward

Can we move the END model forward towards realization of its goals? We believe we can if we can close the knowledge gaps identified above. Some have questioned END's utility due to concerns that it fails to adequately define what is "natural" (Klenk et al. 2008) and that it is largely irrelevant in a changing climate (Kramkowski 2012). While these are valid concerns, there is much that END could achieve if the shortcomings identified above were addressed. The outcomes of any END plan are only as good as the information that goes into it. Thus what is required is a better understanding of natural disturbance regimes and their ecological effects.

The level of information and knowledge required to develop and implement ecosystem management plans to meet specific objectives is often significantly underestimated. Land managers are frequently under political and economic pressure to take action with incomplete knowledge, and while we cannot expect that all knowledge gaps will be filled before taking action, we must remain humble and apply the principles of adaptive management. Land management agencies should work together to create landscape level plans. While we are moving past issues-based management towards more systems-wide forms of planning, there are still significant planning barriers and silos between different management agencies. Often the required experts in fire ecology do not work for the agencies that are doing the management planning. When building END plans, the relevant experts need to be consulted and included to inform managers what we do (or do not) know about the fire regimes for the area in question.

There are some things that we can implement immediately to improve outcomes of END management:

1. Conduct research to define the extent and location of mixed severity fire regimes.
2. Utilize prescribed burning following harvest to restore the chemical effects of fire. This practice was more common in the past, but has been largely abandoned due to clean air regulations in various jurisdictions, and concerns regarding the potential of prescribed fire escapes. With advancements in smoke forecasting, and with significant public relations work, restoring this practice could significantly

improve the END model by using the physical and chemical processes of fire in management.

3. Recognize the different effects of fire versus harvesting on the abundance of different plant functional groups relative to fire (evaders, invaders, avoiders, resisters and endurers). Silviculture usually focuses on planting trees, but seeding with non- tree species (understory shrubs or herbaceous vegetation) could help reduce the differences in community structure between fire and harvesting. This could help preserve ecological community diversity in harvest units and increase ecological resilience.
4. Update growth and yield models and fire spread models to reflect more complex stand structure resulting from mixed severity disturbances.
5. Increase the temporal depth of our understanding of the interaction of disturbance regimes and ecosystem structure. This can be achieved by a variety of means including paleoecological techniques, dendrochronological analysis of dead wood, historical climate modelling, historical photograph analysis to measure landscape change (Stockdale et al. 2015), and modelling, among others.
6. Establish and/or link to existing programs to document the long-term ecological effects of experimental and operational END management on biodiversity, forest productivity, and ecosystem function. While this has been done in some places like the western boreal with the EMEND project (Spence and Volney 1999), the eastern boreal using the TRIAD approach (Messier et al. 2009), the DEMO experiments in the Pacific northwest (Aubry et al. 2004), or research programs out of fRI Research in the northern Alberta foothills (Andison 2000), among others, it is vital to conduct local studies for a given management area and not just make the assumption that nearby areas have the same fire regime. As gaps are identified, they need to be closed in an adaptive management framework.

2.11. Conclusion

If fire regimes are to be used as "coarse filters", we need to determine that the size of the mesh is right. Even with a wide range of views on the subject, and the variable rates at which it is being adopted, the case for adopting the natural disturbance paradigm of forest management continues to be strengthened (Kuuluvainen 2009; Lindenmayer et al. 2012),

but is not without significant challenges regarding acceptance among stakeholders (Andison 2009*a*, 2009*b*, 2010; Clark and Slocombe 2011). It shows considerable promise as a management tool, and can be adapted and used to manage for uncertainty moving forward. Currently, many of the criticisms and shortcomings of the approach are due to a lack of understanding of fire regimes required to implement the END model. With focused research and discussion, we can strengthen the base upon which we build the END model.

It is important to recall the feedbacks required in adaptive management. Further cycling through the learning and structured decision making process is required, and it is clear that there is a need to conduct more research into historic disturbance regimes to improve our ability to emulate natural disturbances and maintain ecological integrity of our managed landscapes.

Chapter 3: Extracting ecological information from oblique angle terrestrial landscape photographs: performance evaluation of the WSL[1] Monoplotting Tool

3.1. Abstract

While aerial photography and satellite imagery are the usual data sources used in remote sensing, land based oblique photographs can also be used to measure ecological change. By using such historical photographs, the time frame for change detection can be extended into the late 1800s and early 1900s, predating the era of aerial imagery by decades. Recent advancements in computing power have enabled the development of techniques for georeferencing oblique angle photographs. The WSL Monoplotting Tool is a new piece of software that opens the door to analyzing such photographs by allowing for extraction of spatially referenced vector data from oblique photographs. A very large repeat photography collection based on the world's largest systematic collection of historical mountain topographic survey images, the Mountain Legacy Project, contains >6,000 high resolution oblique image pairs showing landscape changes in the Rocky Mountains of Alberta between ca. 1900 – today. We used a subset of photographs from this collection to assess the accuracy and utility of the WSL Monoplotting Tool for georeferencing oblique photographs and measuring landscape change. We determined that the tool georeferenced objects to within less than 15m of their real world 3D spatial location, and the displacement of the geographic center of over 121 control points was less than 3m from the real world spatial location. Most of the error in individual object placement was due to the angle of viewing incidence with the ground (i.e., low angle/highly oblique angles resulted in greater horizontal error). Simple rules of control point selection are proposed to reduce georeferencing errors. We further demonstrate a method by which raster data can be rapidly extracted from an image pair to measure changes in vegetation cover over time. This new process permits the rapid evaluation of a large number of images to facilitate landscape scale analysis of oblique imagery.

[1] WSL – Abbreviation for the Swiss Federal Institute for Forest, Snow and Landscape Research

3.2. Introduction

Being able to measure and document ecosystem and landscape change through time contributes to our understanding ecological dynamics, historical ecology, environmental management and ecological restoration (Higgs et al 2014). Numerous studies using widely varied methodologies have shown that across western North America, grasslands and open canopy forests have been lost to encroachment and densification of forests over the last century (Arno and Gruell 1983; Gruell 1983; Baker 1992; Rhemtulla et al 2002). These changes are believed to have caused decreased landscape vegetation diversity (Shinneman et al 2013; Romme et al 2009), and increased the susceptibility of forests to wildfire (Baker 1992; Agee 1998) and biotic disturbances (Arno et al 2000). We need to be able to measure ecosystem change over a relevant time period and spatial resolution to inform management needs and enhance planning efforts designed to mitigate these ecological problems.

Remote sensing can measure changes in patterns, but is limited to the recent past: in most of North America the earliest aerial photography dates to the 1930s, and in the region of western Canada we study, the first systematic aerial survey was conducted in 1949. To look further back in time, some researchers have used historical land based oblique angle photographs (as old as 1870s) to describe ecosystem change. Hastings and Turner (1965), Gruell (1980; 1983), and Webb (1996) have all used historical repeat photography to describe ecological changes over periods of 80-100 years. Some of these studies predated the development of Geographic Information Systems (GIS) software, which limited their ability to conduct spatial analyses. Hastings and Turner (1965) were limited to purely descriptive measures showing changes in the natural vegetation of arid and semi-arid areas. Gruell (1980; 1983) was also limited to using repeat photography for purely descriptive purposes to show the scale of vegetation change in Montana and Idaho between 1871 and 1982. He described in detail the changes in vegetation type, fire frequency, and processes such as snag fall but lacked any quantitative or spatial measurements. Webb (1996), like previous investigators, was able to describe changes in vegetation type, and changes in river flow patterns, but was limited by the inability to spatially reference the images and quantify these changes.

For repeat-photograph studies done using GIS, the challenges associated with georeferencing oblique angle photographs limited the analyses to describing changes qualitatively. Attempts at quantitative assessments of historical photographs by labor intensive manual procedures have been limited to the small spatial scale of only a few

meadows (Roush et al 2007) or single valleys (Rhemtulla et al 2002; Watt-Gremm 2007). Rhemtulla et al (2002) used repeat photography to document 80 years of vegetation change in the montane regions of Jasper National Park, Alberta, Canada. These images are part of the Mountain Legacy Project (Higgs et al, 2009) collection, and image pairs were adjusted to enable accurate overlays. Changes in vegetation type were noted at each pixel, but due to changing scale within the images (foreground pixels representing less area than background pixels) this only revealed relative rather than absolute measures of change. Their assessment was limited to 20 pairs of photographs, covering a total area of approximately 6,000 ha.

Manier and Laven (2002) used remote sensing classification techniques to detect change in vegetation using historical photographs spanning 80-100 years, but their comparisons between the two time periods were restricted to relative changes such as relative percent cover, relative patch size and the number of patches by vegetation type.

Roush et al (2007) used a grid overlay in a GIS to evaluate change in repeat photographs in Glacier National Park, Montana, USA, but the grid was of a constant size overlain on the images. Grid cells over vegetation in the image foreground represented less area than grid cells over vegetation in the background of the image. Their study was detailed in relative change measures, but no spatial measurements were obtained.

Watt-Gremm (2007) used the Mountain Legacy images to examine vegetation change in the Blakiston Valley of Waterton Lakes National Park, Alberta, Canada. In an effort to make the change measurements spatial, he used a GIS to create a spatial grid, drape it on a DEM of the study area, manually rotate the view of the draped grid to match the photo perspective as closely as possible, and overlay it on the image in Photoshop to classify the vegetation in each grid cell. The process used by Watt-Gremm inspired the development of the raster analysis method described below in this paper. However, Watt-Gremm did all of this manually, and thus was limited to analyzing only seven image pairs, covering less than 2,000 ha in total, and the accuracy is unknown because the perspective-rotated grid overlain on the image was not analyzed in a GIS.

As the above-mentioned researchers were beginning to make use of historical photographs, and working on methods to quantify change, they all noted the challenges associated with spatially referencing land based oblique images. Early attempts to create computer based methods for spatially referencing oblique-angle photographs (Aumann and Eder, 1996; Aschenwald et al 2001; Mitishita et al 2004; Corripio 2004; Fluehler et al 2005) have not been widely adopted because they are task-specific, relatively inaccurate, and

were restricted by available computing power. Corripio (2004) developed a computer program that georeferenced oblique images by allowing the user to manually rotate a DEM to a perspective matching an oblique image. The user inserts control points on the image, and visually aligns these control points with the same marks on the DEM. For small areas, and when working with very few photographs, this system functions as intended, however it takes considerable trial and error to get the alignment correct. The image is then rubbersheeted and can be viewed in a GIS. Watt-Gremm (2007), Roush et al (2007), Bozzini et al (2012), and this author have all used the Corripio (2004) application, and while it worked for its intended purpose, all of these researchers found it is very difficult to orient the camera correctly to obtain an accurately georeferenced rubbersheeted image.

Considering the aforementioned challenges associated with quantifying changes visible in terrestrial oblique angle images, no one has yet used historic photographs to conduct a landscape-level (100's-1000's km2) quantitative assessment of vegetation pattern change across a century. With a growing number of historical photograph collections that can be used to measure historical landscape change, developing accurate methods to analyze these images within a GIS would be of great benefit. One such collection is the Mountain Legacy Project (MLP) (Trant et al 2015, Higgs et al 2009), which has collected images in Canada from the 1870's to 1950's. Of nearly 140,000 historical oblique angle photographs, 4,500 photographs have been retaken since the late 1990s, with repeat photography ongoing. The majority of these photopairs are of the landscape of the Alberta Rocky Mountains and foothills region (see Figure 3.1).

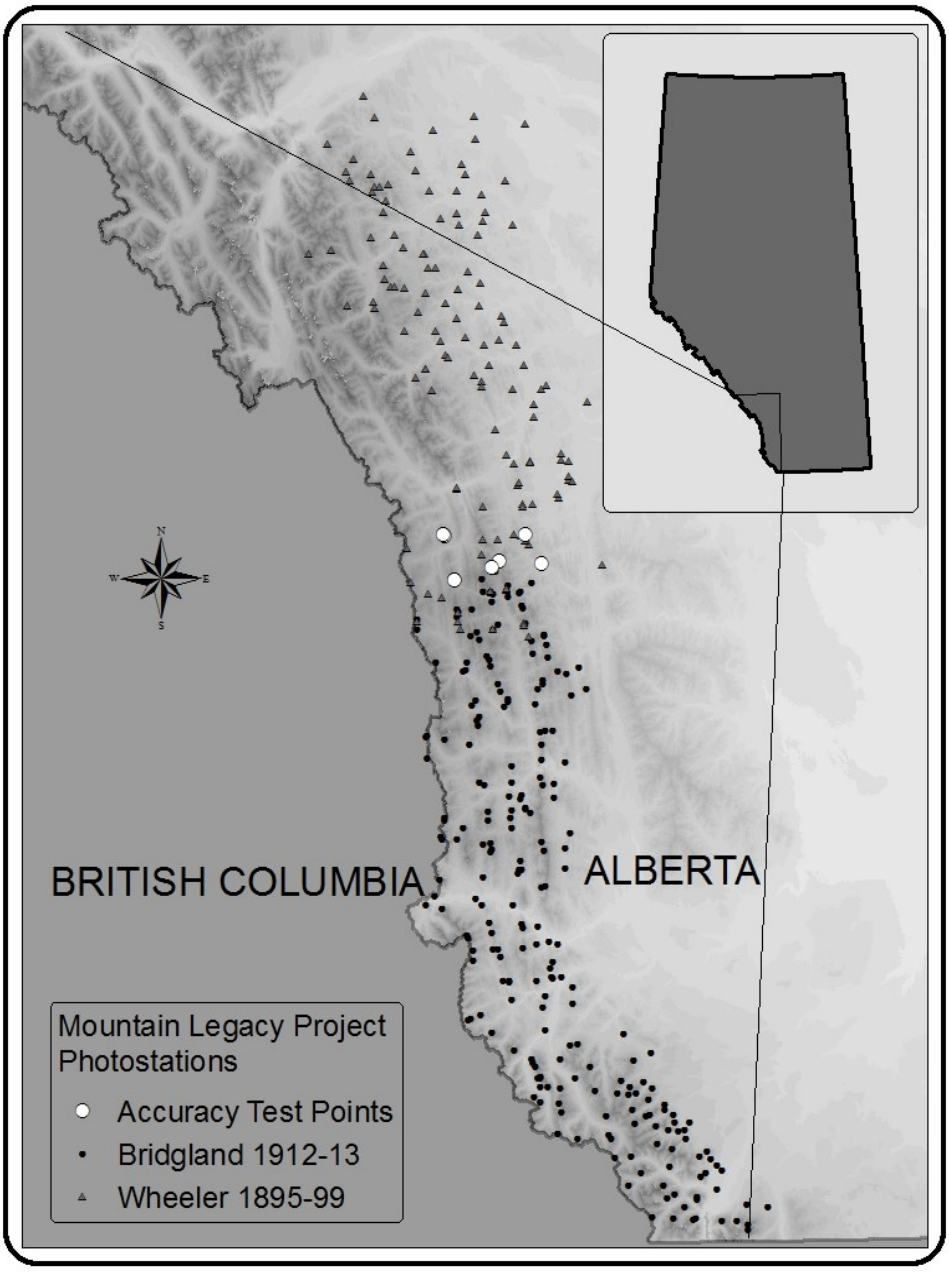


Figure 3.1: Location map of Mountain Legacy Project images in the Wheeler 1895-1899 and Bridgland 1912-1913 surveys. Points used in the assessment of the monoplotting tool were chosen from the overlap area between the two surveys. Eight images were used from the six photostations.

Recently, Bozzini et al (2012) developed a new method of georeferencing oblique photographs to extract vector data: the WSL Monoplotting Tool (WSL being the acronym for the Swiss Federal Institute for Forest, Snow and Landscape Research). This tool has been demonstrated to have utility for evaluating landscape change in mountainous topography (Steiner 2011; Wiesmann et al 2012), but has not been assessed for its accuracy or utility for analyzing large collections of imagery. The WSL Monoplotting Tool is a software tool that relates each photographic pixel to its real-world latitude, longitude, and elevation.

The georeferencing of oblique angle, terrestrial images developed by Bozzini et al (2012) and implemented in the WSL Monoplotting Tool follows the photogrammetric monoplotting procedure, which is described in detail in Aumann and Eder (1996), Strausz (2001), and Steiner (2011). The assumption of monoplotting is that the camera, a point on the photograph in two dimensions (2D), and the corresponding point in the real world in three dimensions (3D) all lie in a straight line. This relationship is visually depicted in Figure 3.2 and described by the collinearity equation below. The equation's variables are shown in Figure 3.2 and described in the following text:

$$x_a - x_0 = -f\,\frac{r_{11}(X_A - X_C) + r_{21}(Y_A - Y_C) + r_{31}(Z_A - Z_C)}{r_{13}(X_A - X_C) + r_{23}(Y_A - Y_C) + r_{33}(Z_A - Z_C)}$$

$$y_a - y_0 = -f\,\frac{r_{12}(X_A - X_C) + r_{22}(Y_A - Y_C) + r_{32}(Z_A - Z_C)}{r_{13}(X_A - X_C) + r_{23}(Y_A - Y_C) + r_{33}(Z_A - Z_C)}$$

The values r_{11}-r_{33} are functions of the rotation angles of the camera about the X, Y and Z axes. The value x_0y_0 is the 2D coordinate of the line drawn from the projection center of the camera through the center of the image, and f is the focal length of the camera.

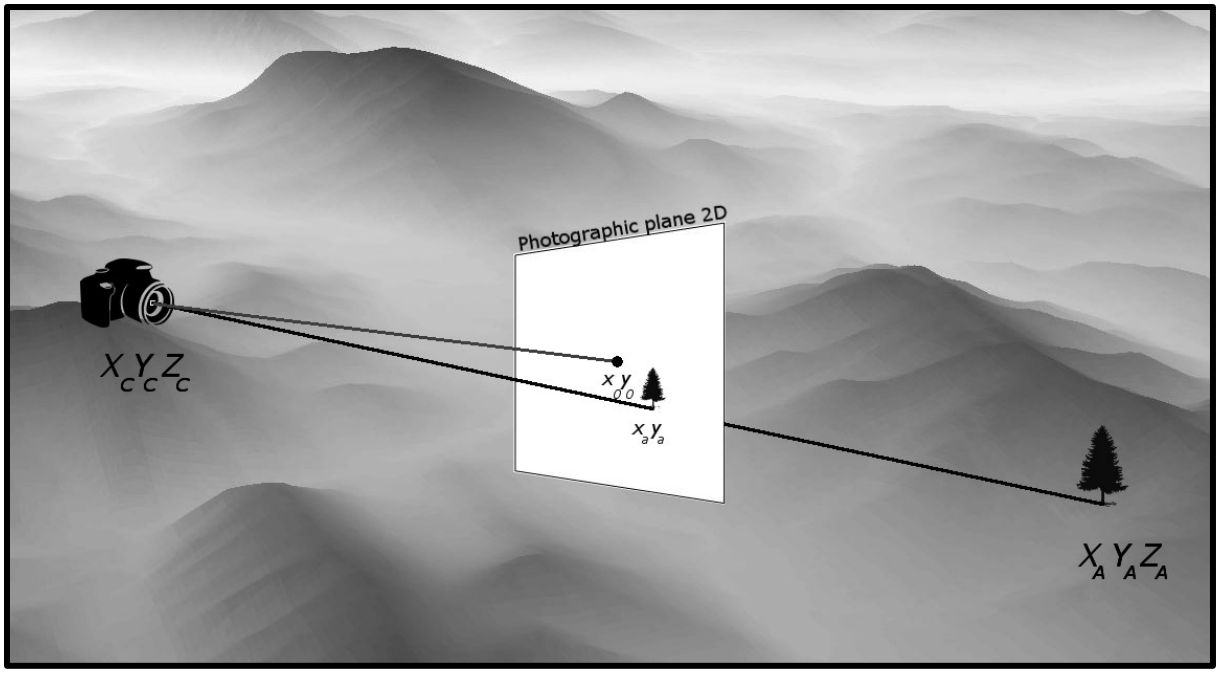


Figure 3.2: The collinearity condition as illustrated by the relationship between the camera, and an object with pixel coordinates $x_a y_a$ in the 2D photographic plane, its real world 3D coordinates $X_A Y_A Z_A$. $X_C Y_C Z_C$ indicates the location of the camera position in 3D space.

The WSL Monoplotting Tool contains a routine which computes the values of the collinearity equation and determines the external (extrinsic) and internal (intrinsic) camera parameters. This routine starts from a set of five or more control points (CPs) whose correspondence between the 2D oblique image and the 3D real world is well known.

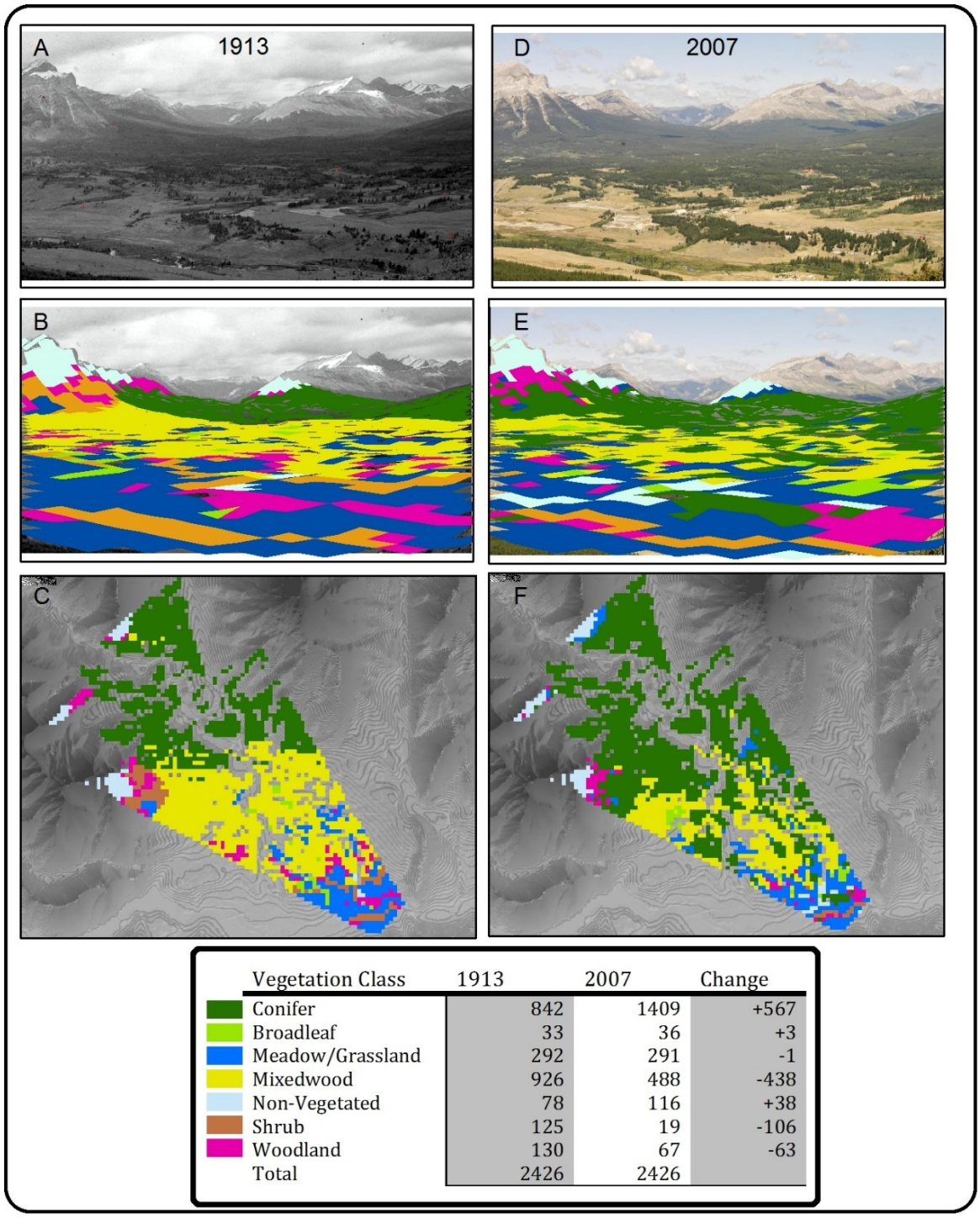
Once the collinearity equation has been solved, vector data (polygon, polyline or point) can be extracted from the image, and exported to a GIS for analysis. In addition to supporting the export of spatially referenced vector data to a GIS, spatially referenced vector data can also be imported to the WSL Monoplotting Tool, and overlain on the photograph. The WSL Monoplotting Tool has been designed for vector data, which is suitable for analysis of a small number of images, or for extracting data from only a portion of an image. To evaluate large landscapes it is considerably more efficient to classify vegetation on a grid or raster basis: by doing so we can take advantage of image classification techniques common in remote sensing. The WSL Monoplotting Tool's utility can be expanded by pairing it with GIS functionality in ArcGIS (or another GIS). We can create a workflow permitting manual or automated image classification (Jean et al 2015) of oblique angle images and to translate the data into a raster format.

Our objectives were:

1) To conduct an assessment of the accuracy of the WSL Monoplotting Tool for georeferencing and extracting vector data from oblique landscape images.
2) Establish a procedure for classifying raster data from oblique photographs using the WSL Monoplotting Tool. While the tool has been designed to extract vector data (polygons, lines and points), manual vegetation classification of images by drawing polygons is a labour intensive process, and there are advantages to using a raster/grid-based classification scheme.

3.3. Methods

The WSL Monoplotting tool requires a photograph(s) to be analyzed, a digital elevation model (DEM) of the area visible in the photograph, and control points (extracted from orthophotos, maps or field data). Often for studies involving oblique photographs the use of field data to establish control points is impractical due to the time and cost required to physically visit the locations contained within the images and measure control points using GPS devices. Thus, we established control points by matching features in the oblique photograph with orthophotos of the area. High resolution bare earth DEMs derived from 1m LIDAR data were made available from the Province of Alberta Ministry of Environment and Sustainable Resource Development (AESRD), as were orthophotos with a resolution of 0.5m per pixel. Because the orthophotos were recent (2005-2008), we used only the recent (repeat) images from the Mountain Legacy Project photograph collection in this assessment of the accuracy of the WSL Monoplotting Tool (to ensure greater accuracy in placement of the control points). Eight images in total were chosen from the Wheeler Irrigation Survey collection (original images were taken between 1895-1899, repeats from between 2007-2009) and the Bridgland 1912-1913 Survey (original images were taken between 1912-1913, repeats from between 2005-2009). The photostation locations for all Wheeler and Bridgland surveys are displayed in Figure 3.1. An example of a historic repeat photo pair is shown in Figure 3.3A and 3.3D.



Vegetation Class	1913	2007	Change
Conifer	842	1409	+567
Broadleaf	33	36	+3
Meadow/Grassland	292	291	-1
Mixedwood	926	488	-438
Non-Vegetated	78	116	+38
Shrub	125	19	-106
Woodland	130	67	-63
Total	2426	2426	

Figure 3.3: Mountain Legacy Project image pair (A, D). The original image is from 1913, and the repeat image from 2007. Panels B and E show the change in vegetation classes in the oblique view, and C and F show the orthogonally transformed view. Each grid cell is 100mx100m (1ha). The attached table shows the total (hectares) of each vegetation category at each time and the total change.

3.3.1. ***Accuracy Test***

From the field data collected by the MLP photography crews, the location of each photostation was recorded in UTM coordinates, and the approximate azimuth for each photograph is known. A two by two array of DEM tiles (each equivalent to a Canadian National Topographic System 1:20000 map sheet) was loaded, and for each image within this extent, the ArcGIS 10.2.2 Viewshed tool was run to determine each image's approximate field of view (FOV). From these images, eight were chosen so that there was minimal overlap between any image FOVs (two images shared a small portion of each other's FOV). These eight MLP images taken in 2009 were selected from six photostation locations (two photostation locations were represented by two images each) (Figure 3.4). Orthophotos supplied by AESRD were taken in 2007, which greater facilitated identification of features for use as control points. Each image was divided into three horizontal segments (foreground, midground, background) and three vertical segments (left, center, right), for a total of nine segments. Within each segment two to three easily identifiable features were identified as control points. These 21-27 control points in each image were established by matching features visible in the oblique angle photograph and in the corresponding orthophoto. Objects chosen were isolated trees, boulders, road intersections, and other easily recognized features. These control points were distributed throughout the image in the fore-, mid- and background, and from one side to the other. Control points were marked on each MLP image in the GNU Image Manipulation Program (GIMP version 2.8.4), and the orthophotos and DEMs were examined in ArcGIS 10.2.2. The UTM coordinates (latitude, longitude and elevation) of each control point were recorded in a table. These control points were used for georeferencing the images, and for testing the accuracy of the georeferenced outputs created by the WSL Monoplotting Tool.

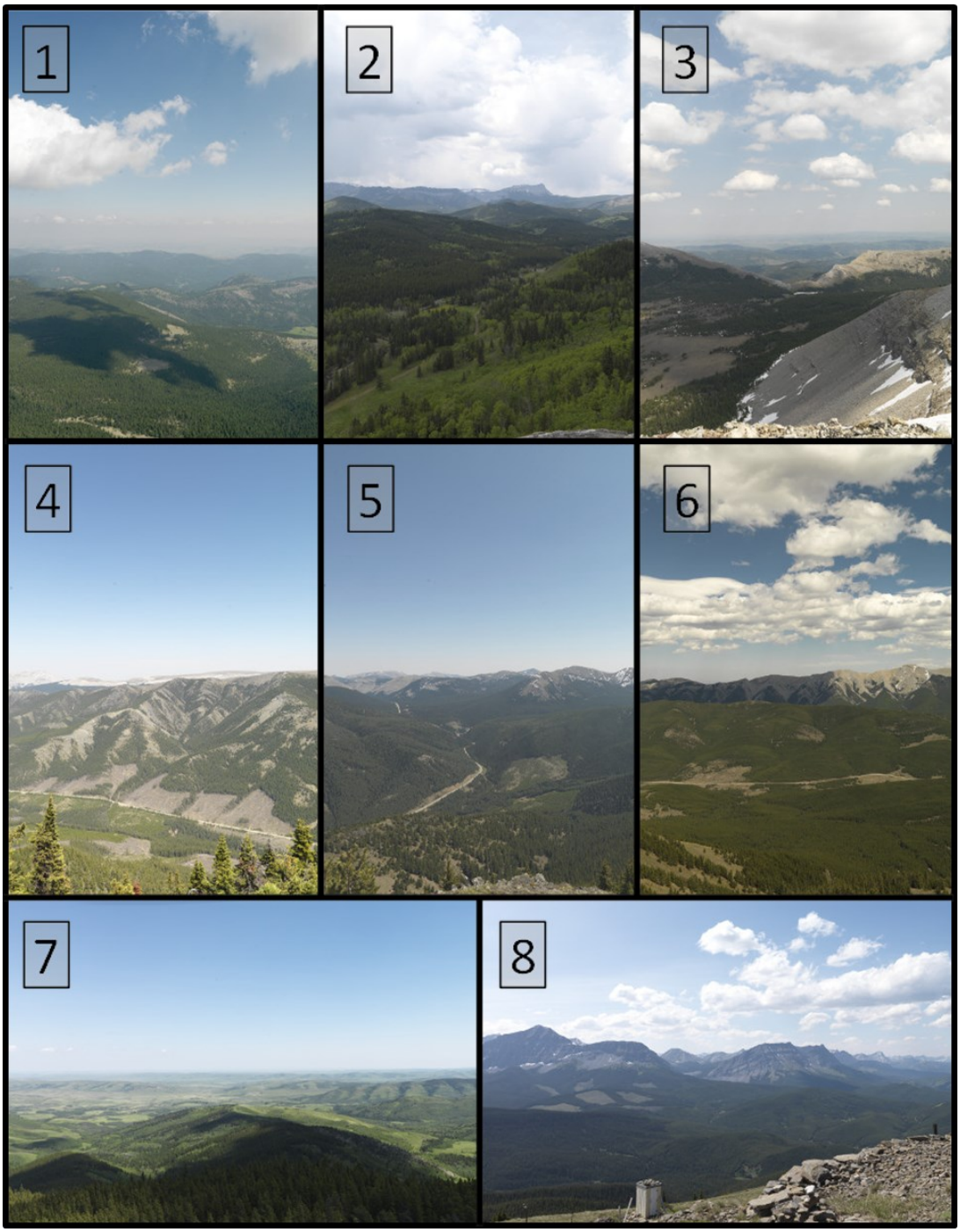


Figure 3.4: Mountain Legacy Project images used in testing the accuracy of the WSL Digital Monoplotting Tool. All images are from Higgs et al (2009).

In theory, if the control points could be precisely placed in both 2D and 3D space, the DEM was a perfect representation of the real world, and there were no lens or sensor/film distortion in the camera, there would be no error in object placement after georeferencing the image. However, DEMs are rarely perfect models of the real world, pixellation in both the

2D images and the orthophotos from which the 3D location is derived cause errors in the placement of features, and camera lenses and sensors/film are rarely perfect.

Figure 3.5 shows how these sources of uncertainty affect the accuracy of object placement using the monoplotting principle. There is nearly always some deviation between the ray from the center of the camera through the points identified by the user on the image (p') and the DEM (r_P'), and the ray that aligns camera, and the real points on the image (p) and in the real world (r_P). If there is no error, the angle between these rays is zero, but this angle of error increases as the points become less accurately placed. The WSL Monoplotting Tool calculates the angle of the deviation between the rays r_P' and r_P.

For each control point $\boldsymbol{P_c\ (p, P)}$ the following data are computed (see Figure 3.5 for each parameter represented pictorially):

O	"Origin", or position of the camera (in real world coordinates), $X_cY_cZ_c$ in the collinearity equation and Figure 2.
p	given point on the image, x_ay_a in collinearity equation and Figure 2.
P	given point in the world (real world coordinates), $X_AY_AZ_A$ in collinearity equation and Figure 2.
π	2D image plane.
π_P	plane through P, perpendicular to the ray r_P.
p'	projection of P on π (pixel point computed from the real world point P) as a result of displacement of P to P'.
P'	reprojection of p on the DEM (real world point computed from the pixel point p) as a result of displacement of p to p'.
P''	projection of P' on π_P
r_P	given ray OpP
r_P'	computed ray $Op'P'$
d	distance between p and p'
R	distance between P and P''
D	distance between P and P'
$\alpha(r_P, r_P')$	angle between r_P and r_P'

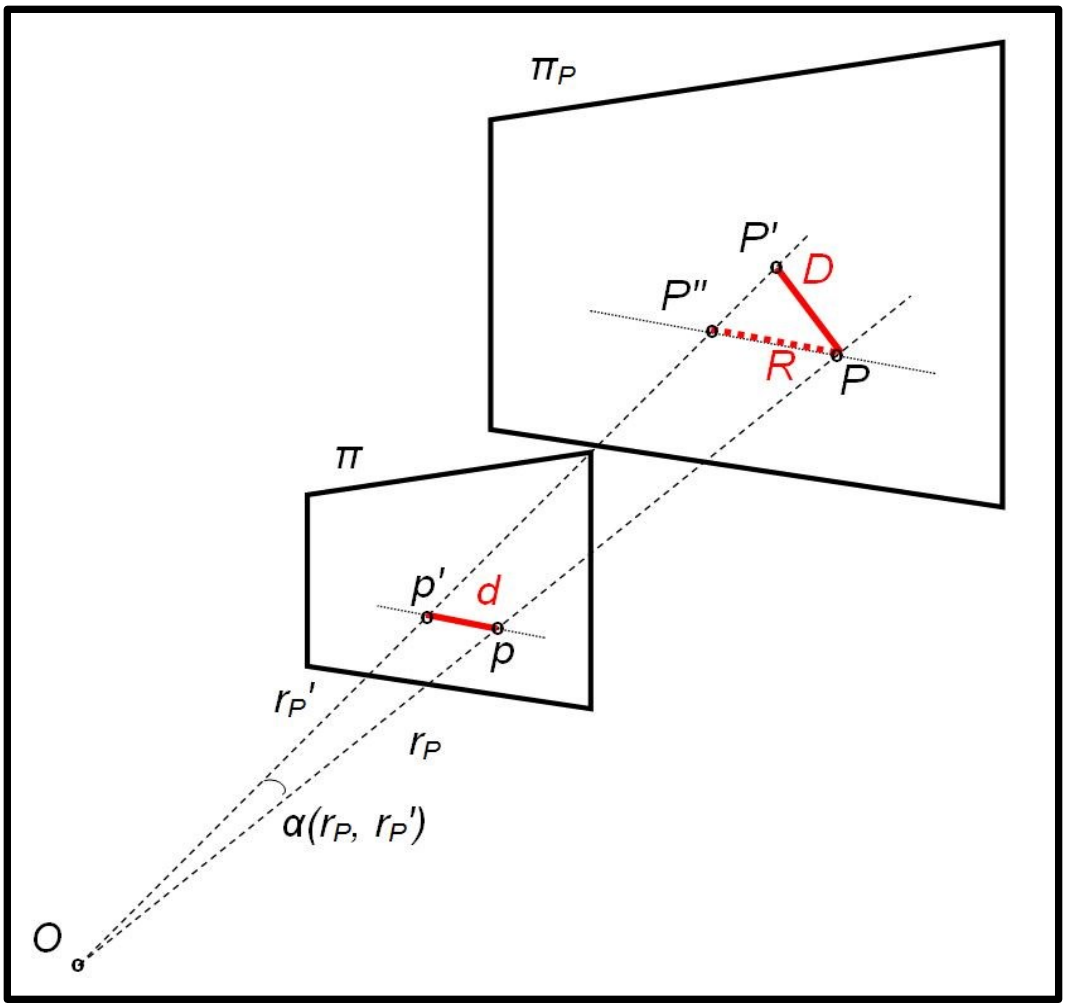


Figure 3.5: The difference between the modeled relationship and the real relationship between objects in the real world (3D) and the photographic plane (2D). Ray O*pP* (r*P*) aligns the camera, the image point *p* , and the real world object *P*. Ray O*p'P'* shows the computed line due to errors arising from control point placement, lens distortion, sensor/film distortion, DEM inaccuracies, or any combination of these factors. If image point *p* is misplaced by the user at *p',* the real world point *P* is then projected to *P'.* Conversely, if the real world point *P* is displaced at *P'*, then image point *p* gets projected at *p'*. Errors in the placement of both *p* and *P* compound the displacement.

For each image to be georeferenced, all 21-27 control points established on that image were initially selected to compute the intrinsic parameters of the camera. Control points used in the computation routine to solve the intrinsic camera parameters are referred to as Registration Points. Once the intrinsic parameters were computed, for practical reasons the three least accurate registration points – as indicated by the "angle error" described above ($\alpha(r_P, r_P')$) – were dropped, and the intrinsic parameters of the camera were

then recalculated using the reduced number of registration points. This procedure was repeated iteratively, dropping the three least accurate registration points, then recomputing the parameters of the camera on the reduced number of registration points until the “best” six registration points remained. For example, if an image had 21 control points identified, we calculated the camera parameters first using 21 registration points, then 18, 15, 12, 9, and finally 6. For each image, these remaining six registration points and the resulting intrinsic camera parameters defined the “best camera” solution. To assess the sensitivity of the WSL Monoplotting Tool to the accuracy of control point placement, we also created “worst camera” and “random dispersed camera” solutions. The least accurate six registration points as defined by $\alpha(r_P, r_P')$ (the first two sets of three control points dropped in the “best camera” solution) were used to create a “worst camera” solution. Additionally, six evenly dispersed semi-randomly chosen registration points were used to create a “random-dispersed” camera solution.

These three different camera solutions (best, worst and random-dispersed) were developed for each image to compare the accuracy of the WSL Monoplotting Tool in reprojecting the remaining control points (ie. non-registration points). The discarded control points from each image’s best-, worst- and random dispersed- camera solutions were then used as test points to determine how accurately they would be reprojected on the DEM using the WSL Monoplotting Point tool. These test points were drawn on the images in the WSL Monoplotting Tool and the spatially referenced points exported to ArcGIS. Some points had to be excluded from the analysis. The excluded points were all situated at the top of hills and ridges with highly oblique angles of viewing incidence (see Figure 3.6) from the observation point to the control point location. The small amounts of error in the angle these object were placed at resulted in the points being displaced either above the horizon (an infinite distance away), or a large distance away on ridges behind the correct one.

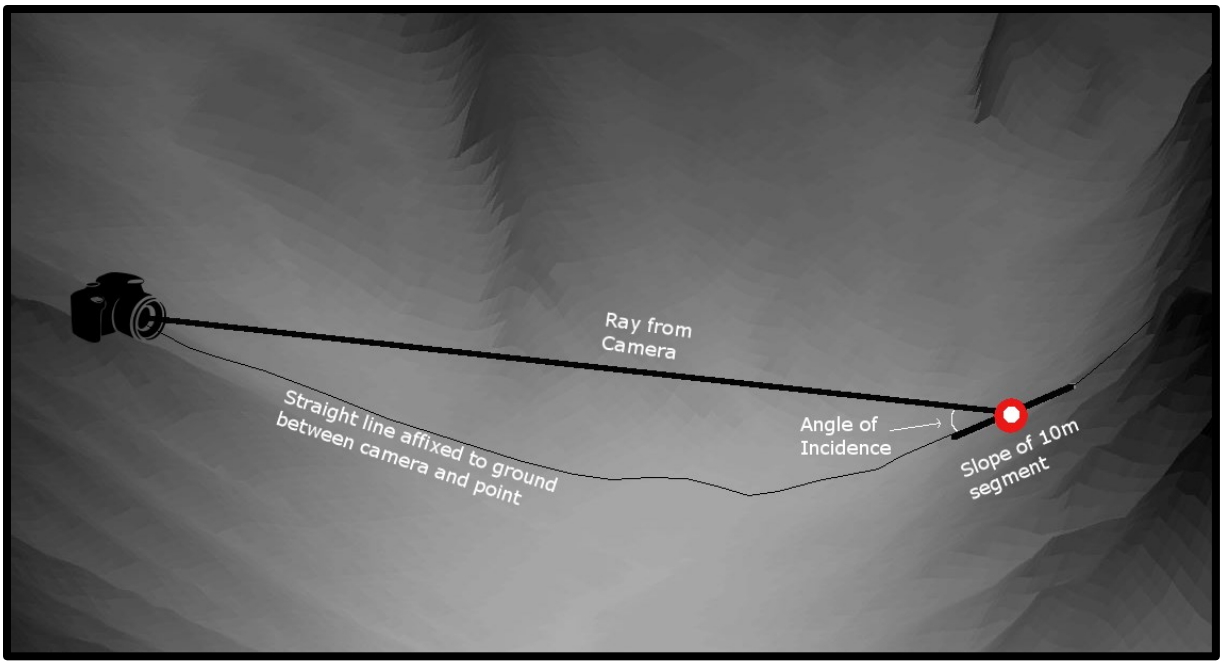


Figure 3.6: The angle of viewing incidence between the ray from the camera (r_P), and the slope of a 10m line segment from a line between the camera and point *P* affixed to the ground. Lower angles of incidence increase the 3D distance between *P* and *P′*.

Several different metrics were calculated to assess the accuracy of the WSL Monoplotting Tool intrinsic camera parameters computed for each image. Table 1 shows all computed values associated with the error testing. These computed values for each image and its best-, worst- and random-dispersed internal camera solutions are as follows:

- *Error vector length* (per point): Parameter *D* in Figure 3.5. The distance between the reprojected Test Point (*P′*) and actual Control Point (P) location was measured by creating new line features in ArcGIS, and computing the length of the line.
- *Mean error vector length* (per image-camera combination): for all test points in each image-intrinsic camera solution, the mean error vector length was calculated (arithmetic mean of all *D* per image-camera combination).
- *Angle of viewing incidence* (per point): using a ray from the camera location to each control point the angle between the viewing vector and the mean slope/aspect of a 10m segment of the line running from the camera through the control point and fixed to the ground. See Figure 3.6.
- *Displacement error* (per image-camera combination): the geometric centre (centroid) of all Test Points (*P′*) and Control Points (*P*) for each image-camera combination was computed, and the difference between these centroids is the Displacement Error for

each image-camera combination. Additionally, the geometric center for all Test Points (*P'*) and Control Points (*P*) was calculated for each camera solution (all images combined) to determine the total landscape displacement error.

To determine whether the mean vector length (errors in object placement) is a function of the angle of incidence, or the distance from the camera a General Linear Model was constructed in SPSS version 21 as:

Vector length error (meters) = Intercept
+ $\beta *$Image (random factor)
+ $\beta *$Distance to camera
+ $\beta *$Angle of viewing incidence.

3.3.2. ***Extracting Raster Data***

To extract raster data from images, a new system was developed and demonstrated using a single image taken in 2006 (Figure 3.7). The image was georeferenced using a best camera solution (as described above). With the X, Y, and Z coordinates of the origin (camera placement) derived from the WSL Monoplotting Tool, a viewshed analysis (ArcGIS 10.2.2) was conducted. The FOV was limited to the horizontal angle of the camera's field of view (parameters Azimuth1 and Azimuth2 of the ArcGIS Viewshed tool) by measuring the azimuths of left and right edges of the image (calculated in the WSL Monoplotting Tool using the Viewshed Parameters export function). The nearest and farthest edge of the FOV (parameters Radius1 and Radius2 of the ArcGIS Viewshed tool) were determined by placing points in the WSL Monoplotting tool at the nearest edge and in the distance at a point beyond which the images become so pixelated that classification of vegetation is difficult. This FOV was calculated using the 1m resolution DEM used in all the above procedures. The Fishnet tool (ArcGIS 10.2.2) was used to create a 100m*100m grid across the landscape contained within the image, and this fishnet grid was intersected with the FOV to isolate the grid cells visible within the images. Cells that had less than 75% visibility were excluded to avoid making assumptions about the content of the invisible portion of the cell. This 100m grid was then transformed from the orthogonal perspective (world coordinates) to the oblique perspective (pixel coordinates) using the WSL Monoplotting Tool's World Shapefile to Pixel Shapefile tool. This perspective-transformed grid was then overlain on the image in ArcGIS. In each grid cell, the vegetation was manually classified as either a) grass, b) shrub, c) open woodland, d) broadleaf, e) mixedwood, or f) coniferous forest cover. The

classified oblique perspective grid cells were transformed back to the orthogonal perspective by using the Pixel Shapefile to World Shapefile tool of the WSL Monoplotting Tool. This fishnet grid was then converted to raster coverage, thereby completing the process of classifying the vegetation visible in the oblique image and converting it to orthogonal perspective raster data ready for summary and analysis in a GIS. This process of georeferencing, overlaying a grid, and interpreting vegetation to a raster coverage was repeated with the original photograph (taken in 1913), and the change in vegetation over 94 years is displayed in Figure 3.3.

3.4. Results

With the available data inputs the WSL Monoplotting Tool had a mean vector length error of 14.7m (s.e. 2.4m) when using the best-camera solution to georeference the image (see Table 1 for all accuracy and error values). The worst-camera solution yielded a mean error vector of 41.2 m (s.e. 4.8m). For the random-dispersed camera solution this measurement error was 24.8m (s.e. 4.5m). When considering the mean displacement error (the geographic centre of all points measured) the measurement error was reduced to 2.9m (best camera), 5.0m (random-dispersed camera), and 9.1m (worst-camera).

Table 3.1: Measurement of errors in monoplotting procedure using eight Mountain Legacy Project images.

Image	Camera Solution	# Registration points	# Test points	Mean Angle Error	Mean Angle of Viewing Incidence (range)	Mean Error Vector length (m) *D* (+/- SE)	Mean Displacement Error (m)
1	Best	6	15	.007	21.3º	4.7 (0.7)	**1.7**
	Random-dispersed	6	15	.027	(9.3 - 32.6)	5.9 (1.0)	**1.9**
	Worst	6	15	.041		14.8 (3.0)	**7.1**
2	Best	6	15	.002	21.3 º	8.3 (1.8)	**1.9**
	Random-dispersed	6	15	.028	(5.7 - 36.4)	18.1 (4.0)	**9.5**
	Worst	6	12	.115		61.6 (12.6)	**55.4**
3	Best	6	15	.007	18.9 º	11.7 (2.6)	**2.0**
	Random-dispersed	6	16	.038	(7.0 – 36.5	10.7 (2.6)	**2.3**
	Worst	6	16	.218		24.8 (3.6)	**11.1**
4	Best	6	15	.013	27.3 º	16.0 (6.2)	**4.8**
	Random-dispersed	6	15	.085	(4.9 – 48.6)	25.8 (7.6)	**6.9**
	Worst	6	14	.199		10.5 (2.4)	**6.3**
5	Best	6	16	.003	25.8 º	10.2 (3.0)	**4.3**
	Random-dispersed	6	14	.034	(3.6 – 47.9)	12.3 (4.2)	**7.8**
	Worst	6	16	.067		72.1 (16.7)	**38.7**
6	Best	6	18	.003	28.3 º	7.8 (1.6)	**1.9**
	Random-dispersed	6	19	.016	(2.4 – 51.8)	9.3 (1.8)	**0.8**
	Worst	6	19	.074		23.3 (3.9)	**16.9**
7	Best	6	15	.014	14.7 º	48.0 (14.9)	**26.4**
	Random-dispersed	6	15	.220	(1.9 - 38.3)	108.5 (27.0)	**35.1**
	Worst	6	14	.338		96.2 (24.6)	**70.4**
8	Best	6	12	.007	20.1 º	11.9 (3.5)	**9.2**
	Random-dispersed	6	12	.034	(13.0 – 43.3)	8.7 (2.5)	**7.0**
	Worst	6	11	.129		33.4 (8.4)	**27.0**
Total	Best		121			14.7 (2.4)	**2.9**
	Random-dispersed		121			24.8 (4.5)	**5.0**
	Worst		117			41.2 (4.8)	**9.1**

The effect of distance to camera and angle of viewing incidence variables on the vector length error were significant at $\alpha = 0.05$. Using all 121 test points, the General Linear Model yielded the equation:

$$\begin{aligned}\text{Error vector length (meters)} = 12.1 &+ \beta * \text{Image} \\ &+ 0.003 * (\text{Distance to camera}) \\ &- 0.596 * (\text{Angle of viewing incidence}).\end{aligned}$$

Using the methods described above to extract raster data from images in the WSL Monoplotting Tool, Figure 3.7 shows an image a) that has been georeferenced using the WSL Monoplotting Tool, b) a viewshed calculated, c) a spatial grid intersected with the viewshed and d) transformed to the oblique perspective and overlain on the image, e) the vegetation in the image classified, and f) retransformed back to the orthogonal perspective for analysis in a GIS. Figure 3.3 shows the same image (3.3D), its original paired image (3.3A) from 1913, the classified vegetation in both images (3.3B, C, E, F), and a summary of the changes in vegetation cover between the two time periods.

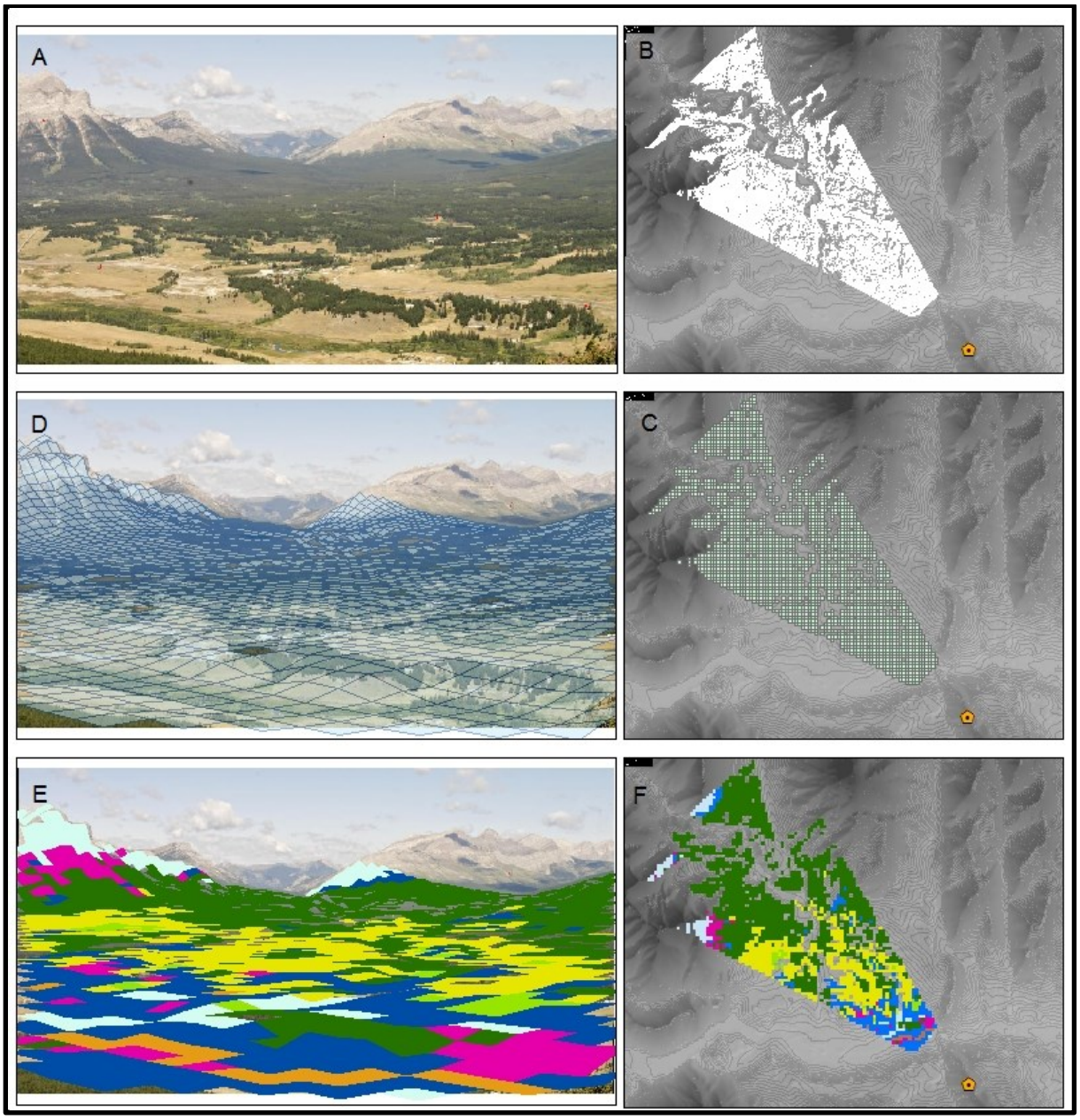


Figure 3.7. Procedure for raster analysis of oblique images using the WSL Monoplotting Tool. A) Image to be analyzed, B) Viewshed of image after georeferencing to identify photo origin, C) 100mX100m fishnet grid intersection with the viewshed, D) oblique perspective of fishnet grid overlain on image, E) classified vegetation on image F) orthogonal transformation and spatially referenced classified grid cells.2 column image.

3.5. Discussion

The Mountain Legacy Project notwithstanding, there are many historical repeat photograph collections available showing landscape change in many regions of the world. Literature and internet searches of the terms “historical repeat photography” yield a large number of different studies, collections, and publications. However, as has been described in the introduction of this paper, these studies have been primarily limited to qualitative or relative comparisons of change (Hastings and Turner 1965; Gruell 1980; Gruell 1983; Webb 1996). In the limited number of studies that have managed to spatially evaluate changes, they were limited to studying very small areas of the landscape due to the complexity of the analytical methods available (Rhemtulla et al 2002; Corripio 2004; Watt-Gremm 2007; Roush 2007), and the accuracy of the spatial data outputs is unknown. There are a few studies that have used this new WSL Monoplotting Tool (Steiner 2011; Wiesmann et al 2012; Bozzini et al 2012), but they have only extracted vector data. While these are spatially accurate, they too have only looked at limited spatial scales. To be of value in management applications, imagery showing change is most useful when:

- it can be described and quantified
- is spatial and accurate
- assessment procedures are rapid to facilitate landscape-scale analysis.

In this paper, we have not only used the WSL Monoplotting Tool to georeference and extract classified vector data to assess the spatial accuracy of the tool, we have developed a new approach to extract raster data from oblique angle images. This raster-based approach will permit researchers to evaluate large image collections, and has the potential to be combined with automated image classification techniques common in the field of remote sensing. This adds considerable value to the multitude of repeat photography projects that have been conducted, are in progress today, or will be conducted in the future. These new techniques create the potential to expand the field of remote sensing to a much wider user audience and array of data sources. Oblique-angle land-based imagery is a widely available data source, but until recently has not been useable for quantitative spatial analysis. While repeat land-based photography has been used as a data source in many assessments of landscape change, researchers have been restricted to qualitative or relative comparisons of change over time, or have been restricted to quantitative analysis of very small areas.

The value of historical repeat photography to document ecological change has long been recognized (Pickard 2002; Webb et al 2010), primarily because it extends the temporal scale we can study. Aerial photography is temporally limited to the early to mid-20th century, whereas land based photographs exist into the late 19th century. While extending the temporal record by roughly 50 years may seem trivial for some lines of inquiry, it extends our window of photographic observation to the beginning of the European settlement era in western North America, South America, New Zealand and Australia, and other parts of the world. In all parts of the world, the beginning of the 20th century was a time when considerable change was occurring on the landscape, as this coincided with a period of rapid population growth and technological advancement. While it is our objective to evaluate landscape scale vegetation change using historical photography, there are many other potential uses of historical imagery. Paired historical photographs can show changes in glaciation, river channels, shorelines, erosion, land use, architecture, settlements, and many other things.

The value in being able to spatially quantify things visible in oblique angle terrestrial photographs is not only restricted to studying historical change predating the era of aerial photography. Even when the temporal period of interest is covered by aerial imagery, it is often not readily accessible, and can be expensive to acquire new imagery. Land based photographs are ubiquitous and considerably less expensive to obtain. Furthermore, terrestrial based oblique angle imagery can be useful without being paired to historical imagery. Provided the data inputs outlined in this paper are available (a DEM and some control points), any photograph can be georeferenced and spatial data can be extracted from it.

As we have demonstrated, the WSL Monoplotting Tool is effective for georeferencing oblique angle photographs. With the built in functionality to georeference the image and import and export spatial data, and with tools designed to work with ArcGIS (Viewshed Parameters) and other GIS software, the WSL Monoplotting Tool allows users to accurately and rapidly analyze many images. It is currently being used by the authors of this paper to evaluate landscape level change visible in roughly 150 photo pairs from the Mountain Legacy Project covering approximately 350,000 ha.

The accuracy of the WSL Monoplotting Tool is limited by the mensuration or placement of the control points on the image itself and on the DEM, which is largely a function of the resolution of the images at hand (both the photograph being analyzed and the orthophotos being used to find the control point locations). It can also be limited by

inaccuracies in the DEM, or lens distortion of the camera itself. The influence of distance to the camera on vector length error factor is mainly due to the difficulty in accurately placing control points on the image when they are farther from the camera (due to the high degree of pixelation that occurs when zooming in). Caution should be exercised when interpreting features that are at low angles of incidence and close to terrain breaks, where slight errors in angles can result in large horizontal displacement. Control points should not be placed in the following locations:

- On surfaces that have a very low angle of viewing incidence,
- On tops of hills/ridges/terrain breaks where there is a risk they could be displaced by very long distances.
- Further away than the most distance objects to be classified (i.e., if limiting analysis to within 5km of the camera, control points should be placed within 5km).

While the WSL Monoplotting Tool was not designed to extract or work with raster data, we have herein demonstrated that a workflow can be created that expands the functionality of the tool. The advantage to interpreting vegetation as raster rather than polygon is that a larger number of images can be analyzed, and larger landscape level inferences can be made. The resolution of the raster grid size should be considerably greater than the error in georeferencing accuracy to ensure that classification of the grid cell is spatially correct. If the recommendations regarding control point placement are followed, the errors in georeferencing can be minimized considerably.

With the growing interest in using historical photography, and with recent advancements in computing power, the WSL Monoplotting Tool, in conjunction with GIS software, high resolution DEMs and orthophotos can be used to accurately georeference and classify land based photographs to document and quantify ecological change over a longer time period than that afforded by aerial imagery.

3.6. Acknowledgements

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Chapter 4: A century of landscape change in the southern Rocky Mountains and Foothills of Alberta

4.1. Abstract

We used 137 historical repeat oblique photography pairs from the Mountain Legacy Project (originals taken in 1913, repeat images in 2008) to quantify vegetation change over an area of 320,000 ha in the southern Rocky Mountains of Alberta, Canada since the beginning of the 20th century. We developed a new method to overlay a spatially referenced 1-ha grid on the photographs, classified the vegetation into seven distinct vegetation types (closed canopy conifer-, broadleaf deciduous-, or mixedwood-forest, open canopy woodlands, shrublands, grasslands and meadows, non-vegetated), and then assessed vegetation change between the two time periods. We were also interested in identifying which factors could help explain variation in vegetation change across the landscape, such as elevation, solar insolation (slope and aspect), and disturbance history (fire, harvesting, and other anthropogenic disturbances). Therefore in the photographs we also classified visible wildfire, timber harvesting, or anthropogenic disturbances on the landscape in both time periods, and supplemented these disturbance observations with provincial timber harvesting and wildfire records. We found that closed canopy coniferous-, broadleaf deciduous-, and mixedwood- forests have increased on an area basis by 35%, 45% and 80% respectively relative to a century ago, while concomitantly the area covered by grasslands and open canopy woodlands declined by 25% and 39% respectively. Only 9% of the landscape was in an earlier successional state ("reverse succession"), while 28% was in a more advanced successional state in 2008 as compared to 1909. More than 87% of the area that was in an earlier successional state in 2008 than 1909 occurred in areas that had known disturbances (timber harvesting, wildfire, anthropogenic disturbance) between the two time periods. The majority of the change has occurred in the Montane Natural Subregion, and in the Subalpine Natural Subregion (42% and 26% respectively in a more advanced state). The loss of open canopy woodlands is acute across the entire landscape, with the exception of the Foothills Fescue Natural Subregion. Grassland and meadow losses are most acute in the Subalpine and Alpine Natural Subregions. We found that there was an increased probability of vegetation change to a more advanced succession condition at higher elevations across the study area and in areas receiving lower amounts of solar insolation, which coincide with northerly aspects (cooler and moister). We also found that as time since fire or harvest increased, a given site was more likely to have returned to (or gone

past) its pre-disturbance successional stage. The changes observed are consistent with what we would expect to see due to lengthening of fire return intervals.

4.2. Introduction

Wildfire is a disturbance that exerts control of vegetation structure and interacts with variation in topography, climate, fuels (vegetation) and anthropogenic factors to drive patterns of vegetation composition across the landscape (Parisien et al. 2006). Since the arrival of European settlers in western North America in the late 19th and early 20th centuries we have seen considerable change in the fire regimes and resulting vegetation composition of much of the landscape (Arno 1980; Barrett 1996; Bradley and Wallace 1996; Heyerdahl et al. 2001; Wright and Agee 2004; Rogeau 2005b; Van Wagner et al. 2006; Romme et al. 2009). If this landscape is to be managed in an ecologically sustainable fashion, improved understanding is required of the natural range of variability of vegetation composition, and the magnitude and causes of the changes in disturbance regimes and landscape level vegetation composition over time (Stockdale et al. 2016).

Many studies have shown lengthening of fire return intervals (the time between fire events at a point in space) and fire cycles (the amount of time required to burn an area equal in size to a given area of interest) in much of the intermountain west of North America (Hawkes 1979; Tande 1979; Barrett 1996; Rogeau 1999; Heyerdahl et al. 2008 Rogeau 2016). A review of some early studies by Arno (1980) showed that pre-1900 fire return intervals were between 15-30 years in the Montane Natural Subregion, and from 30-150 years in the Subalpine. We have seen these increase dramatically over the past 100 years as there has been little fire throughout the region since the early 20th century (Tymstra et al. 2005; Van Wagner et al. 2006; Heyerdahl et al. 2008).

Vegetation change across the landscape has accompanied the lengthening of fire return intervals. Studies in the Rocky Mountains of the northern USA and Canada have shown substantial increases in closed canopy forest cover since the turn of the 20th century at the expense of grasslands and open canopy woodlands (Strong 1977; Gruell 1983; Campbell et al. 1994; Brown et al. 1999; Rhemtulla et al. 2002; Fulé et al. 2002; Hessburg et al. 2005; Higgs et al. 2009). Arno and Gruell (1983) described a largescale conversion from grasslands in favour of conifers from 1936-1981 in southwestern Montana. Gruell (1983) also showed forest encroachment on former grasslands, and substantial increases in canopy closure within the forested areas between 1871 and 1982 in Montana and Idaho.

Rhemtulla et al. (2002) found that grassland cover in the montane valleys of Jasper National Park had declined by 50% between 1915 and 1998. Watt-Gremm (2007) found that forest cover had nearly doubled, with subsequent loss of shrublands and grassland in Waterton Lakes National Park between 1913 and 2006. Strong (1977) compared the vegetation composition between the late 1800s and the 1970s and found that what was considered "aspen parkland proper" in the 1970s used to be "groveland", and what was "groveland" in the 1970s had previously been fescue grasslands. Campbell et al. (1994) showed a biome-wide replacement of grasslands by aspen dating to the 1880s and 1890s along the margin of what is now referred to as the aspen parkland all the way from Regina, through to Edmonton and south to Calgary. While there has been considerably less research on this topic in Canada as compared to the United States, the patterns appear to be consistent; where there used to be abundant grassland and open forest habitat for diverse communities of plants and the animals, there is now contiguous, closed canopy, lower-diversity mature forest that is at high risk to loss by fire (Moore et al. 2004, Fulé et al. 2004) and forest diseases and insects, such as the Mountain Pine Beetle (Dordel et al. 2008, Hughes et al. 2006).

The Government of Alberta (through the Ministry of Agriculture and Forestry, and the Ministry of Environment and Parks), Government of Canada, and the forest industry have implemented management plans throughout the Alberta Rocky Mountains that recognize these landscape changes (White et al. 2003; Government of Alberta 2007; Walkinshaw 2008; Government of Alberta 2010). While these stakeholders all recognize numerous ecological issues that have arisen as a result of shifting disturbance regimes, and have prioritized recovery of these ecosystems within their natural range of variability, there is a significant gap in our understanding of how much change there has been, how patterns of change varied across the landscape, and which factors explain this variation. Without this understanding, the problem is not adequately quantified and there are no measurable targets for restoration activities.

Disturbance regimes and historical vegetation change are typically studied using a variety of techniques: a) dendroecology to document fire history (Barrett and Arno, 1988; Marcoux et al. 2015; Chavardes and Daniels, 2016) and stand dynamics (Ehle and Baker 2003; Axelson et al. 2009); b) paleoecology to analyse lake sediment cores for relative and absolute charcoal abundance over time (Carcaillet et al. 2001; Campbell and Campbell 2000) and pollen to describe past vegetation composition (MacDonald et al. 1991; Lorenz 2009; Prichard et al. 2009); c) examination of historical records from fire occurrence

databases (Bergeron et al. 2001; Tymstra et al. 2005; Parisien et al. 2006), or maps (Johnson and Fryer 1987) and aerial imagery to document changes in vegetation (Fichera et al. 2012; Chuvieco 1999; Pickell et al. 2013; Rutherford et al 2008); d) modelled using inputs derived from these sources of information (Li, 2000; Wimberly and Kennedy, 2008; Rogeau 2016). Furthermore, the interactions between vegetation composition and disturbance regime relationships are frequently modeled (Keane et al. 2004; Calkin et al. 2005; Wimberly and Kennedy 2008).

One largely untapped data source to examine landscape vegetation change is using historical repeat photographs, which are taken from the same place, showing the same area at two or more points in time. Many historical repeat photography studies have shown forest invasion of grasslands across a wide geographic area of western North America between the late 1880s and early 1900s to the present day (Hastings and Turner 1965; Gruell 1983; Arno and Gruell 1983; Webb 1996). The primary drawback of these studies is the observations were not spatially quantified. By the late 1990s researchers began to examine historical repeat photographs to provide quantifiable measures of change. Rhemtulla was able to make relative change measurements by using overlays of historic and present day images, but she was unable to say how much absolute area had been lost. Manier and Laven (2002) in Colorado, and Roush et al. (2007) in Glacier National Park in Montana were also limited to relative change measurements. Watt-Gremm (2007) pioneered a method described in Stockdale et al. (2015) overlaying a spatially referenced grid on oblique photographs, however his observations were limited to seven image pairs covering less than 2,000 ha in total, and he could not determine the accuracy of his methods due to technological constraints. Chapter 3 (Stockdale et al. 2015) provided a review of the latest techniques in oblique angle image analysis and developed a new method to enable rapid and accurate assessment of a large number of historical repeat photographs to enable studies of landscape level ecological change.

The Mountain Legacy Project (Higgs et al. 2009; Trant et al. 2015) is a repeat photography project larger than any other similar project in the world with more than 120,000 historical images taken in the mountainous regions of western Canadian by numerous topographic map surveyors dating from the late nineteenth to the early twentieth centuries. Survey crews climbed peaks, ridges and promontories throughout the Canadian Rocky Mountains and usually took a series of photographs from each station to create a panorama of what was visible from that location (MacLaren et al. 2005) from which they developed topographic maps. To date, more than 6,000 of these images have been

repeated (in the southern Rockies from 2005-present day) from the exact original locations ("paired images") which provide us a view of the changes on the landscape since the turn of the 20th century. While some researchers have used these images to examine ecological change in focused locations (Rhemtulla et al. 2002; Watt-Gremm 2007; Kubian 2013), their real potential to evaluate large spatial scale ecological change throughout the Alberta Rocky Mountain region has remained largely untapped.

The southern Alberta Rocky Mountain region in the centuries prior to the European settlement period can be examined from the perspective of human history to gain an understanding of how anthropogenic influences may have changed the landscape itself. First Nations people in the region belonged to the Blackfoot tribe, a semi-nomadic people whose primary food source was bison (Brink 2008). The Blackfoot people hunted bison in numerous ways, including the use of buffalo jumps which were numerous throughout the region (Brink 2008). By the mid-1700s, with the introduction of horses and guns, their way of life and methods of hunting began to fundamentally change (Brink 2008). The first European explorers (David Thompson, Peter Fidler, and others) came through the region in the late 1700s and early 1800s, which led to a period of active trade between the Blackfoot and Europeans, and ultimately the signing of treaties with the Canadian government in the 1870s; this effectively removed the Blackfoot people from the land and moved them onto reservations (Brink 2008). This was followed by extirpation of the bison by 1880, the building of the railroad by 1897, and the founding of numerous European settlements in the region (Brink 2008). The Mountain Legacy Project images provide a clear snapshot of what the landscape looked like at the end of this period of tumultuous change.

The principal objectives of this study were to quantify the changes in vegetation composition since European settlement in the early 20th century in the southern Alberta Rocky Mountains and foothills, and to determine what factors might help explain variations in patterns of change across the landscape. From what previous studies in the area and nearby regions suggest, we hypothesized that the following changes in vegetation would be observed in the southern Rocky Mountain and foothills region of Alberta:

- Vegetation in 2008 is further along in succession than it was at the time of European settlement in the late 1800s and early 1900s, and will vary due to the influences of:
    a. Elevation (which affects temperature and vegetation composition)
    b. Solar insolation (which affects temperature, moisture balance, and is driven by slope and aspect)

c. Disturbance history of the landscape as measured by time since fire, time since timber harvest (which affect the length of time vegetation has to recover from destructive disturbances)
d. Anthropogenic land use such as agriculture, settlements, and clearing related to roads, powerlines, railways, pipelines and other rights-of-way.

- Grasslands and open canopy woodlands have been lost due to forest encroachment between the settlement period and the present day, and the degree of change will vary due to the same influences described above.
- Grasslands and open canopy woodlands that are being lost to forest encroachment are being lost in proximity to historically extant forest stands due to the forest progressively expanding from extant edges rather than seeding in the open areas unattached to historical forest.

4.3. Methods

4.3.1. *Study Area*

We used the Bridgland 1913-1914 survey of the Crowsnest Forest Reserve and Waterton Lakes National Park to define the study area. These photographs were taken from 236 unique photographic locations ("stations"), and covered the area of the Rocky Mountains from the Alberta-BC border to the Porcupine Hills, and from the USA-Canada border to Sentinel Pass (142km to the north). The total study area was defined as the smoothed perimeter of the area visible in the selected photographs (see description in section 4.3.1.1) capturing all the visible landscape. This study area is a landscape 87 km north to south, 52km east to west, and encompasses 318,300 ha (see Figure 4.1). It is located in the southern Rocky Mountains of Alberta with the Continental Divide (Alberta – British Columbia border) along the western edge, Turtle Mountain (UTM north 5487345) at the southern edge, the western slopes of the Porcupine Hills (UTM east 715919) along the east, and Sentinel Mountain (UTM north 5575605) as the northern limit. The elevation ranges from 1114m above sea level in the southeastern corner to 3094m above sea level. The area is largely mountainous to the west, with vast areas of open grassland in the southeastern portion of the study area. The Natural Subregions of the study area are Alpine, Subalpine, Montane, Fescue Grasslands and Foothills Parkland (Natural Regions Committee, 2006). See Figure 4.1 and Table 4.1 for the total area and the percentage of the total study area occupied by each Natural Subregion.

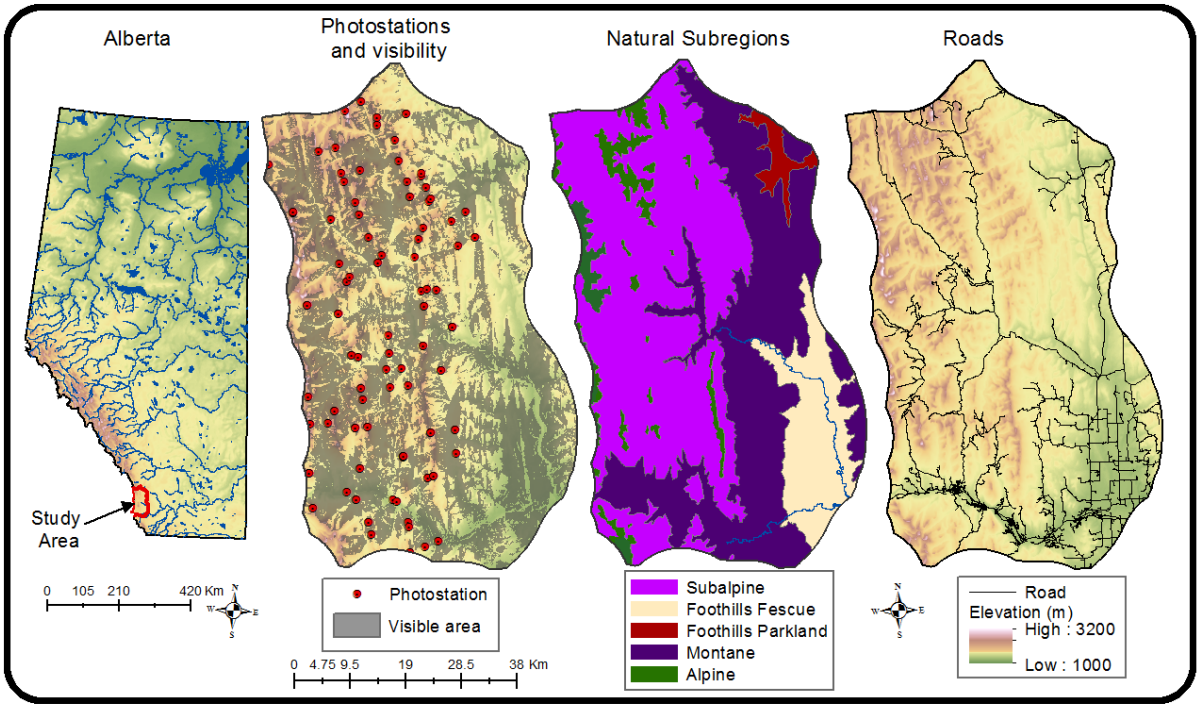

Figure 4.1: The first map panel shows the location of the study area in the southwestern corner of the province of Alberta. The second panel zooms in on the study area and shows the locations of photostations from the Bridgland 1913-1914 survey used in the study with the total visible area of the landscape from 137 paired photographs indicated. The third panel shows the Natural Subregions of the area, and the final panel shows elevation and the locations of the roads in the study area. The dense cluster of roads in the southern part of the study area coincides with locations of towns and settlements.

Table 4.1: Distribution of Natural Subregions across the study area, and the portion of the total landscape and Natural Subregion (Natural Regions Committee 2006) visible in the selected photographs.

Natural Subregion	Study Area (ha)	% of Total Study Area	Visible Area (ha)	% of Visible Area	Visible as % of study area
Alpine	14,468	4.5	9,058	5.0	62.6
Subalpine	144,611	45.4	86,092	47.2	59.5
Montane	117,091	36.8	59,636	32.7	50.9
Foothills Fescue	35,990	11.3	25,282	13.8	70.2
Foothills Parkland	6,140	1.9	2,515	1.4	40.1
TOTAL	318,300	100	182,583	100	57.4

4.3.1.1. Image Selection

Paired historical and modern photographs of the Bridgland 1913-1914 Crowsnest survey (Higgs et al. 2009) from the Mountain Legacy Project were used to measure landscape vegetation change across the study area (see Figure 4.2 for example photographs). Based on the assessment of the WSL Monoplotting Tool by Stockdale et al. (2015, Chapter 3) which suggested that the highest level of vegetation detail could be derived by restricting analysis of the area in photographs within 5 km of the camera location we overlaid a 5km grid across the entire study area. In each grid cell we pooled all photographs taken from any station within that cell and chose 2 images at random. In the event that a grid cell did not have any stations located in it or had only one image available, we chose an extra image or two from adjacent grid cells whose fields of view faced towards the grid cell that lacked a station. From this procedure we selected a total of 137 images pairs to be georeferenced and classified.

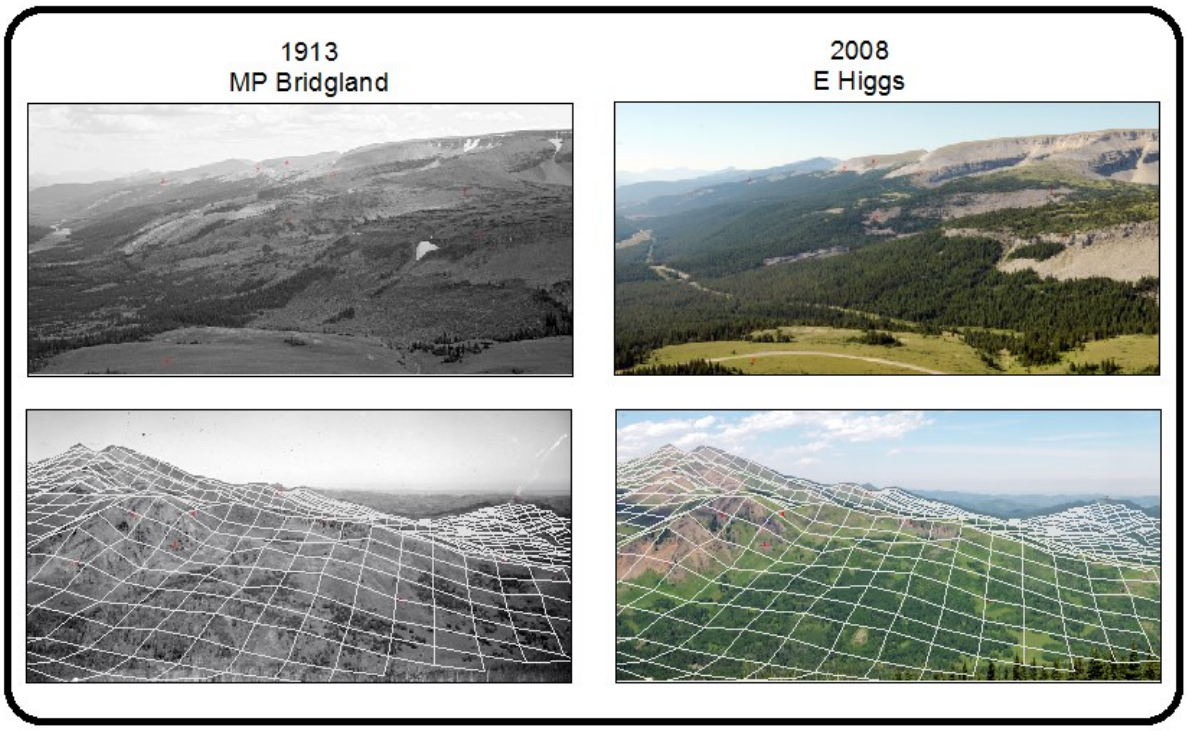

Figure 4.2: Mountain Legacy Project paired photographs from the 1913-1914 Bridgland Survey repeated in 2008 by Higgs. The second row shows an image pair with a georeferenced 1 ha grid overlaid. This overlay grid was used to classify vegetation within each cell to measure change between the two time periods.

To assess the landscape coverage of the selected images we calculated the viewshed (ArcGIS 10.2.2 Viewshed tool) for each image to a distance of 5km from the camera. This required: a) determining the station location; and b) measuring the azimuth of both vertical edges of a photograph. The station locations were provided by the MLP Crew Field Notes. While the azimuth was recorded for most images by the MLP Field Crew, we discovered that the field crews had been inconsistent with setting the declination on their compass in the field, and the recorded azimuths were unreliable. We therefore determined the azimuth using ArcGIS to create line segments on high resolution orthophotos that matched the edges of features visible along both the left and right side of each photograph, measuring their compass bearing, and then computing the mean bearing of these two lines (the image center).

From the resulting viewshed analysis we determined that many photographs shared large portions of their viewshed with other photographs, but there were several large holes in the total coverage. We discarded photographs with the most overlap, chose a new one at random from the same (or adjacent) station, and recomputed the total viewshed of all images iteratively until we had maximized the total landscape coverage of the photographs chosen for the study at 57.4% of the total study area (see Table 4.1). To ensure that the visible landscape was a representative sample of the total study area we compared: a) the visible portion of the landscape in each Natural Subregion against the proportional area distribution of Natural Subregions across the study area (see Table 4.1); and b) the distribution of solar insolation (as a single variable proxy for slope and aspect) in the total study area as compared to the visible area (see Figure 4.3). See below for further details on how solar insolation was calculated. The overall distribution of each Natural Subregion was fairly consistent between the visible landscape and the study area, however, the Montane Natural Subregion was slightly underrepresented and the Foothills Fescue was overrepresented (50.9% of the total area of Montane visible, and 70.2% of Foothills Fescue visible compared to the global mean visibility of 57.4%) (Table 4.1). The overall distribution of solar insolation was very similar based on visual interpretation of the overlap between frequency histograms showing the distribution of solar insolation values in the total study area and the visible area (Figure 4.3).

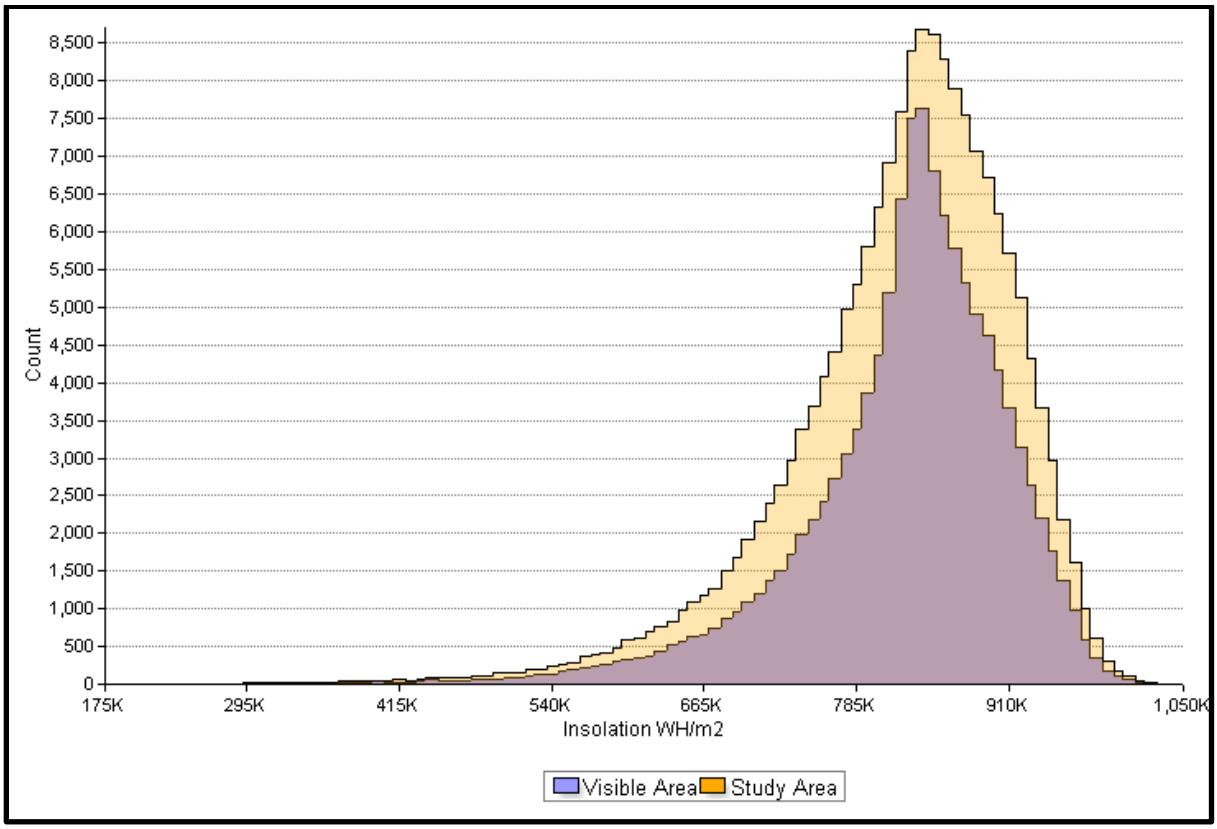

Figure 4.3: Frequency histogram of solar insolation in the total study area (orange curve) and in the visible area (purple curve). "Count" on the y-axis indicates the number of 1-ha cells. The similarity in shape indicates that the visible area is representative of the total study area.

4.3.2. ***Image Georeferencing, Vegetation Classification and Disturbances***

We used the WSL Monoplotting Tool (Bozzini et al. 2012) along with the raster data extraction procedure outlined in Stockdale et al. (2015) to georeference and classify vegetation in each repeat photograph pair. In each image, 6-10 control points were placed on the original (1913-1914) and repeat (2008) images by overlaying the image pairs in the Gnu Image Manipulation Program version 2.8 (GIMP), identifying common features between the image pairs, and matching these features in modern orthophotos (taken between 2005-2008) to provide real world Universal Transverse Mercator (UTM) X and Y coordinates. We derived elevation for each control point from a 1-m resolution digital elevation model (DEM) provided by the Province of Alberta Ministry of Environment and Sustainable Resource Development. We georeferenced images, calculated their viewsheds, and intersected a 1 ha spatially referenced grid with the viewshed to yield an "image visible grid". The viewshed was restricted to the distance from the camera station at which we could confidently discern

vegetation categories (described below). We transformed this image visible grid from the orthogonal view to the oblique view and overlaid it on the image to be classified (see example in Figure 4.2).

We classified the vegetation in each visible grid cell as one of seven vegetation types: nonvegetated (NV), meadow and grassland (MG), shrubland (SH), open canopy woodland (WD), broadleaf deciduous (BD), mixedwood (MX), or conifer (CF). Grid cells were classified based on the vegetation category covering the largest portion of the cell, with some exceptions described below. Nonvegetated areas included water, rock faces and outcrops, talus slopes, landslides, sandbars in rivers, and areas of anthropogenic disturbance such as roads, trails, buildings, and other infrastructure that had no vegetation. Meadow and grassland areas included natural grasslands, alpine meadows, farmed fields, grazing lands, grassy yards in developed areas, sedges and rushes in marshy areas, grass growing under power lines, and all other vegetated areas without woody plants. Areas with abundant cover of recently burned and dead trees (resulting from the 1910 fire) that had no visible foliage left on them were assumed to have grass growing in the understory, and these areas were classified as meadows and grasslands. Shrublands were areas that ranged from 20%-100% woody shrub cover with meadows and grassland making up the rest of the site. Shrubs included sagebrush, willow, bog-birch, juniper, and young aspen stands of the same height as these other shrub types (to a maximum of approximately 3 m in height). Open canopy woodland vegetation was defined as areas with distinct space between the crowns of trees, or with distinct patches of trees (i.e. aspen copses, clusters of limber or whitebark pine), regardless of tree species, with either grassland or non-vegetated areas making up the rest of the cell. Tree cover had to be at least 10% to be classified as “open canopy woodland”, up to a maximum of 60% cover, beyond which areas were classified as one of the three closed canopy forest types described below. Broadleaf deciduous vegetation included all broadleaf deciduous trees that were taller than shrub height, and included aspen, poplar, birch, and any cultivated broadleaf deciduous trees that may have been growing in the developed areas of the landscape. Mixedwoods were defined as areas with between 20-80% cover of conifer trees with broadleaf deciduous making up the rest of the cover. If cells had less than 20% conifer cover they were classified as broadleaf deciduous, and if they had greater than 80% conifer cover they were classified as coniferous. Coniferous forests included any coniferous forest species, and were classified as such regardless of their height class, so long as we could recognize the vegetation as coniferous trees and saplings.

To increase the accuracy of the vegetation classification, the original and repeat photos for each photo pair were examined together by overlaying them in GIMP: grasslands, coniferous forests, mixedwood forests and other vegetation types were identified by turning on and off the repeat/original layers to examine the colour, texture, pattern, and context of each grid cell (see Table 4.2 for photo interpretation descriptions used to classify vegetation, and Appendix B for examples of each vegetation type in the original and repeat photographs). Where possible, difficult-to-classify areas of the landscape were examined from other photographs that showed the same location from a different angle and distance. In cases where it was not clear (due to photographic quality, extreme shadow, or other uncertainty) what vegetation category a particular cell should be classified as in either the original or repeat photo, these grid cells were removed from the analysis and not considered any further. Images were analyzed by zooming in and out to increase the accuracy of classification.

Vegetation classification was independent of disturbance attributes, which were also identified in visible grid cells (see below). After we classified each image pair, the visible grid cells were added to a cumulative “total area assessed” layer. For every subsequent image analyzed, we subtracted the “total area assessed” from its “image visible grid” so that each visible grid cell on the landscape was only assessed in a single image. In total the 137 image pairs assessed in this study covered 182,583 ha. The range in the area visible in the image pairs was from 74 ha to 12,725 ha (mean = 1332.7 ha, standard deviation = 1777.8). We developed two vegetation layers (one each for 1913 and 2008), hereafter referred to as Veg1913 and Veg2008.

We also wanted to capture disturbance visible in the photo pairs. In both the original and repeat photographs, recently burned forests were evident. Large fires had burned through the landscape in 1910 (extensively across the study area), and in 2003 (restricted to the very southern edge of our study area). Grid cells in which evidence of fire was visible were classified as “disturbed fire” (DF). The vegetation classifications in these DF cells were MG if most trees were killed, WD if patchy trees survived, and BD, MX or CF if most trees survived (depending on the structure of the snags). If the fire had burned at either low enough severity so as to not cause visible overstory mortality, or at such high severity so as to have burned all dead wood away completely, there was no evidence apparent in the 1913 images, so these would have been missed. In order to better describe the vegetation of the historical landscape prior to the 1910 fires, we rolled back vegetation categories from 1913 to the year before the fire (1909). For grid cells with code “DF”, the pre-fire 1909 vegetation

category was determined by the density and form of the standing dead timber. Grid cells in which dense coniferous snags were visible were classified as CF, those with low density snags were classified as WD, mixed BD and CF snags as MX, and BD snags as BD. This created a third time layer of 1909 (layer Veg1909).

In the Veg2008 layer grid cells with evidence of anthropogenic disturbance from development such as roads, buildings, settlements, powerlines and rail lines were classified as “disturbed developed” (DD) and the vegetation category was assigned based on the dominant vegetation form or labelled as NV if the cell was fully developed (road, house, building, gravel). An additional code was assigned (AG) for agricultural land used for crops. AG and DD codes were combined in subsequent analysis into “anthropogenic disturbance” class (AD).

The subsequent analyses compare the Veg1909 to the Veg2008 layers rather than Veg1913 to Veg2008.Given that the majority of the landscape in 1913 that burned in the 1910 fire recovered quickly to forest again, we did not want to capture the very short lived transient post-fire picture of the landscape that Veg1913 would have provided.

Table 4.2: Description of photo signatures used to classify vegetation in historic black and white, and repeat color photographs. Note that texture and pattern varied considerably due to the distance from the camera. Also see Figures B.1 - B.3.

Vegetation	Historic Color	Repeat Color	Texture	Pattern	Context
Non-vegetated (NV)	Pale grey	Variable by object	Fine to coarse	Highly variable from large slabs to talus slopes peppered with rocky outcrops.	Larger areas extending from mountain tops or from river edges. Confirmed by examining repeat photos and orthophotos.
Meadow / Grass (MG)	Pale grey	Tan, pale green	Smooth	Uniform	Easily identified and isolated from adjacent forest vegetation. Boundaries with NV usually determined by using repeat photos and orthophotos to confirm.
Shrub (SH)	Pale grey (darker than MG)	Pale to dark green, darker than MG	Fine to coarse	Clumpy to uniform	Separated from MG by darker tones and height differences. Separated from CF, MX, or BD by lighter shades and shorter. Common along upper treeline boundary, riparian zones, swales, and depressions.
Woodland (WD)	As MG with dark patches and spots	As MG with dark green patches and spots	Rough	Clumpy to spotty with no more than 60% tree cover	Near to camera very easily distinguished, at greater distances usually separated by distinct two-tone nature.
Broadleaf Deciduous (BD)	Medium grey	Darker green than MG, lighter than CF	Fine to coarse	Uniform with up to 20% darker CF	Near to camera identifiable by color and rounded crown shape. Further from camera separated from MG by darker tone, large height difference; separated from CF by paler tone and rounded shapes.
Mixedwood (MX)	Medium grey mixed with dark grey/black	Mixture of BD and CF shades	Fine to coarse	Range of 20-80% CF cover mixed with BD	Mostly in areas with both pure broadleaf deciduous and pure conifer stands nearby. Differentiated from WD by having no distinct height differences between the light and dark patches, and by presence of pure BD and CF nearby.
Conifer (CF)	Dark grey to black	Medium to dark green	Fine to coarse	Uniform color with up to 20% patchy BD cover, or up to 40% MG cover	Near to camera easy to identify, frequently cover nearly full visible landscape. Separated from MG, SH or BD vegetation by being much darker color.

4.3.3. *Vegetation Change and Transitions*

We created a single polygon layer with attributes for Veg1909, Veg1913, and Veg2008. We measured the total amount of the landscape in each vegetation class in 1909 and 2008, created a transition matrix and tallied all unique (7*7 = 49) vegetation transitions. We assumed a successional pathway for the seven vegetation categories, the sequence being NV-MG-SH-WD-BD-MX-CF (Archibald et al. 1996). For cells that were MG or WD in Veg1909 and underwent succession, we were particularly interested in how much they had changed as these particular vegetation types are the ones most widely believed to have been lost over time. For these (Veg1909 = MG or WD) cells we created ordinal transition values to describe the degree to which they had changed (Table 4.3).

Table 4.3: Coding for the degree of vegetation class change used as the response variable for ordinal logistic regression to detect differences between 1909 and 2008 for areas that were grasslands (MG) or open canopy woodlands (WD) in 1909. -1 indicates "reverse succession", 0 indicates no change, and 1, 2 or 3 indicate progressively greater degrees of change to later successional conditions.

Veg2008	Veg1909	
	MG	WD
Meadow / Grass (MG)	0	-1
Shrub (SH)	1	-1
Woodland (WD)	1	0
Broadleaf Deciduous (BD)	2	1
Mixedwood (MX)	2	2
Conifer (CF)	3	2

4.3.4. *Other Data Layers*

In addition to attributes recorded directly from the oblique angle image pairs, several other data sources were used to test our hypotheses regarding the impact of fires, harvesting, anthropogenic disturbance and solar radiation. Linear features (roads, cut lines, trails, railways, pipelines), timber harvest records, and fire polygons from 1931-2008 were obtained from the Alberta Government official spatial warehouse records.

The data regarding fires from the provincial databases were filtered to exclude fires smaller than 1ha in size as the smaller fires in this area were mostly “extinguished campfires”. Fires greater than 1ha in size only occurred in the 1930s, 1940s and the early 2000s. These fires were all combined with the cells we had coded DF in the Veg1913 layer to create a fire history for the landscape between 1909 to 2008. Forest harvesting records indicated that logging in the area began in the 1950s, however it is possible there are older cutblocks than this that are not in the official government records, nor would these data include forest clearing for other purposes such as agricultural clearing, expansion of communities and other activities.

We buffered linear features (highways and railways by 300m on both sides, all other linear features by 100m both sides) to include areas within these thresholds to account for ecological edge effects caused by the presence of these features. The linear buffers were merged with the AD coded cells developed from the Veg2008 layer to create a single coverage of anthropogenic disturbance. These linear buffers, fires, and harvesting are all shown in Figure 4.4.

To control for potential spatial autocorrelation in the data in the subsequent statistical analyses, watersheds were constructed to serve as a random block effect. Watersheds were delineated using the Hydrology tools in Spatial Analyst (ArcGIS 10.4) and a 1m DEM resampled to 20m. We filled holes in the DEM and calculated a flow direction raster from which we derived a flow accumulation raster. A threshold value of 10,000m^2 was arbitrarily chosen to begin creating first order streams, and stream orders were determined using the Strahler method whereby the order of a stream only increases if it joins with a stream of equal or higher order (i.e. 1st order stream joining 1st order stream = 2nd order stream. 2nd order stream being joined by 1st order stream remains as 2nd order, two 2nd order streams joining become a 3rd order stream, etc.). Pour points were manually placed where second order streams joined third order streams and this created 78 watersheds in the study area. Some watersheds contained no or very few visible landscape grid cells; if there were fewer than 500 cells in any given watershed the visible landscape grid cells in that watershed were reassigned to the nearest watershed containing a larger number of visible landscape grid cells. After reassigning some grid cells, 57 total watershed-blocks were created with a low of 527 cells to a high of 19,482 per watershed; mean watershed size was 5,973 ha (s.d. 5,776) and the range was 1,064 ha to 31,924 ha (Figure 4.4).

To account for the effect of slope and aspect, we used solar insolation as it combines the influence of both variables in a single continuous predictor. We used the Area Solar

Radiation tool in ArcMap 10.4 to calculate mean hourly solar radiation (watt hours per square meter (WH/m^2)) on a 20m digital elevation model, and then resampled the solar radiation layer to a 100m resolution to match the resolution of other data inputs. We used maps from the Government of Alberta Ministry of Agriculture and Forestry that show the mean start (April 24; Julian day 115) and end (October 8; Julian day 282) dates of the growing season (>=5 degrees Celsius) in the Crowsnest Pass (Government of Alberta, 2016) between 1971-2000. This solar radiation layer was also used to test whether the visible landscape was a representative sample of the study area as described above. The solar radiation layer is shown in Figure 4.4.

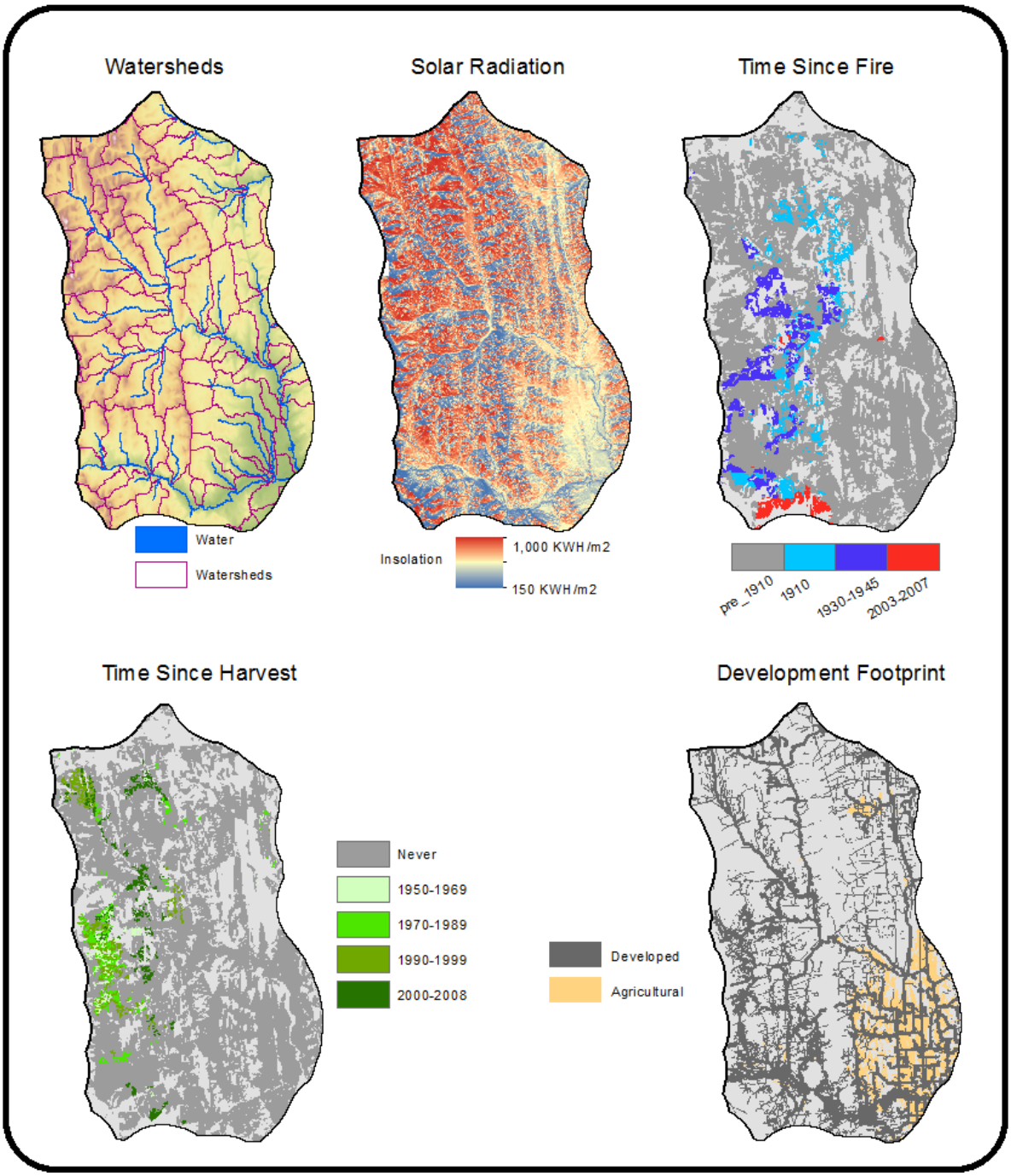

Figure 4.4: Spatial data layers accounting for watersheds, solar insolation, time since fire (TSF), time since harvest (TSH), and the development footprint of the landscape (AD, which combines agricultural and anthropogenic development) . The palest grey background colour in the TSF and TSH maps indicates areas that were not visible in the images we selected for analysis.

4.3.5. ***Determining Factors Contributing to Spatial Variability in Vegetation Changes***

4.3.5.1. <u>Overall Successional Change</u>

To examine the influence of several predictor variables on successional change between Veg1909 to Veg2008 we created a response variable called "successional change" between the two periods based on the vegetation class ("same" (0), "positive" (+1), or "reverse" (-1)). To determine the effect of Natural Subregion on successional change we used a Chi-square test to the null hypothesis that there was no difference in the distribution of "successional change" values by Natural Subregions. To examine whether vegetation change transitions were clustered, scattered, or random, we tested the null hypothesis that successional change was spatially randomly distributed on the landscape by calculating Moran's I statistic using the Spatial Autocorrelation (Moran's I) tool in ArcGIS 10.4. We used the ordinal logistic regression to examine the relationship between the successional change and continuous predictor variables mean solar radiation (solar) and elevation (elev), and categorical variables time since fire (TSF), time since harvest (TSH), and anthropogenic disturbance (AD, including all cells intersecting linear buffers, development, settlements, and agricultural lands). All visible grid cells were used in this analysis (n = 182,583), and were nested within watershed (blockshed) (included as a random block effect) to account for spatial autocorrelation.

TSF was coded as S (short for fires occurring between 2000-2008), M (medium for fires between 1930-1945), L (long for fires in 1910), or XL (extra long for everywhere else on the landscape, as it is very likely that every part of the landscape has burned at some point in history). TSH was coded as XS (extra short for harvesting 2000-2008), S (short for harvesting in 1990-1999), M (medium for harvesting 1970-1989), L (long for harvesting 1950-1969) and N (never for everywhere else on the landscape). The anthropogenic disturbance variable (AD) was coded as the presence (1) or absence (0) of anthropogenic disturbance, which included all cells in the linear feature buffers, agricultural lands, or human settlements. Elevation and solar variables were standardized to z-scores.

We used the ordinal logistic regression procedure CLMM in the R package Ordinal to compare all 5, 4, 3, 2, and 1 variable combinations of TSF, TSH, AD, elev and solar as fixed predictor variables with blockshed as a random factor. We also ran an ecological null model that only included the blockshed random factor and a dummy variable with no variation. The best model was chosen by calculating the Akaike Information Criterion (AIC) for each model

as well as the Akaike Weight (AIC_w), the model having the lowest AIC value was selected as the best model. We computed the Spearman's *rho* statistic to examine correlations between all pairs of predictor variables included in the best model to determine how much these predictors might confound each other. Categorical variables were set as ordered factors to enable this test. We also examined all of the reverse succession on the landscape to see how much of this could be explained by known disturbances (fire, harvesting, and anthropogenic disturbance) using ArcGIS to intersect all reverse succession cells with the know disturbance cells and creating a summary table.

4.3.5.2. Changes from Grassland and Woodland

To determine the effect of Natural Subregion on the degree of successional change in grasslands and woodlands we used two Chi-square tests to test the null hypothesis that there was no difference in the distribution of successional change values among Natural Subregions. These two Chi-square tests were conducted on separate subsets of the full data set, which were filtered for Veg1909 values of MG and WD, respectively. To determine whether grasslands were being lost adjacent to historical forest edges, we converted the vegetation transition raster layer into polygons to group all "like" transitions. These polygons were recoded to create the following categories: "grassland lost"; "grassland gained"; "grassland retained"; "1909 Later"; and "1909 Earlier". The "1909 Later" category included all successional stages later than grassland (SH, WD, MX, BD, and CF), and the "1909 Earlier" category only had NV. We measured all unique boundaries between "grassland lost" and these other types to determine how much of the edge abutted other vegetation types (i.e. length of "grassland lost" edge bordering SH, length of "grassland lost" edge bordering WD, etc.). We also summarized the area of lost grassland adjacent to these other types. This was repeated with the open canopy woodlands, except that the "1909 Later" category included only MX, BD, and CF, and "1909 Earlier" included NV, MG and SH.

To examine factors explaining variation in patterns of change from grassland or woodlands we again used ordinal logistic regression with the same predictor variables described above (without TSH, as there had been no harvesting in the former grasslands or open woodlands), and the same statistical procedures, and model selection criteria. The response variable was the degree of vegetation change, coded to indicate reverse succession (-1), no change (0), and forward change (values of 1 to 3 to indicate the degree of change, see Table 4.3 for full description of these changes); we excluded grid cells that

had undergone “reverse succession” to NV. We ran separate analyses for open woodland (vegetation class WD) and grasslands (vegetation class MG) in each case including only the subset of grid cells that were classified as WD or MG in Veg1909. We tested for spatial autocorrelation in the response variable using Spatial Autocorrelation (Moran’s I) tool in ArcGIS 10.4.

4.4. Results

4.4.1. *Landscape Vegetation Change*

Grasslands occupied most of the landscape in 1909, and coniferous forest was the next most common (Figure 4.5, Table 4.4). All other vegetation categories combined occupied less than a quarter of the landscape, and in diminishing order these were woodlands, non-vegetated, broadleaf deciduous, shrubs, and mixedwoods. In 2008 the vegetation cover was dominated by coniferous forests, with grasslands the next most common vegetation cover. The remaining vegetation categories only occupied less than 21% of the landscape, and these were broadleaf deciduous, non-vegetated, mixedwood, woodland and shrubs in diminishing proportional cover (Table 4.4). The majority of the total landscape was in the same vegetation category in 2008 as it was in 1909 (Table 4.5). More than a quarter of the landscape was in a more advanced successional state, and very little had undergone “reverse succession” (Table 4.5).

Nearly 15% of the visible landscape changed from grassland/meadow (MG) and open canopy woodland (WD) in 1909 to closed canopy forest (BD, MX, CF) 99 years later (Table 4.6, Figure 4.6). While shrubland changed the most in terms of proportion, it only occupied a small part of the landscape to begin with. These relative changes were not uniform across the landscape, with the Montane Natural Subregion having the most area undergoing forward succession, followed in diminishing order by Foothills Parkland, Subalpine, Alpine, and Foothills Fescue. There was a significant difference in the amount of successional change among Natural Subregions with a probability less than 0.0001 of the variation being due to chance (Pearson’s Chi-squared = 16982, degrees of freedom = 8). The summary of transitions between 1909-2008 for all visible grid cells (Table 4.6) shows that the change from meadows and grasslands into coniferous forests was the single largest transition category, with a substantial amount also shifting from grasslands to the other closed forest types (mixedwood or broadleaf deciduous forest). For areas remaining the same, most of the landscape that was coniferous forest in 1909 remained as such in 2008

(a total of 30% of the landscape) and more than half of the grasslands in 1909 were still in a grassland state in 2008 (Table 4.6).

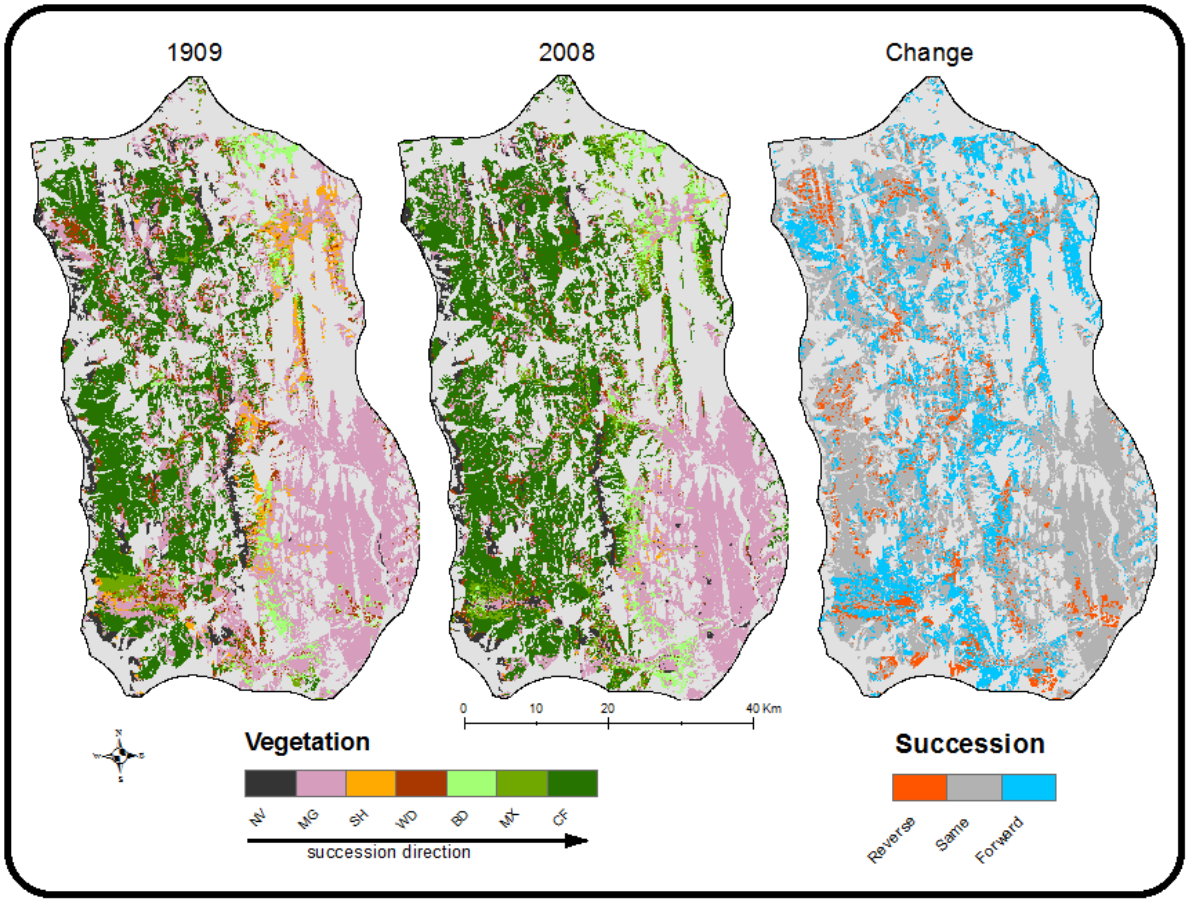

Figure 4.5: Vegetation across the landscape in 1909 and 2008 as measured from 137 historical repeat photography pairs. The vegetation categories are nonvegetated (NV), meadows and grassland (MG), shrubland (SH), open canopy woodland (WD), broadleaf deciduous (BD), mixedwood (MX), and coniferous (CF). The successional sequence of these vegetation types is indicated, as is the overall direction of successional change on the landscape. “Reverse” indicates that in 2008 the vegetation at a given location is at an earlier successional state than it was in 1909, “same” indicates it has not changed, and “forward” indicates that in 2008 the vegetation is in a more advanced successional state than it was in 1909. The pale grey background colour in each map indicates areas that were not visible in the images we selected for analysis.

Table 4.4: Total and percent area for each vegetation category in 1909 and 2008 and the precent change over time. See also Fig. 4.5.

Vegetation	1909 ha (%)	2008 ha (%)	Change from 1909 – 2008 ha (%)
Non-vegetated (NV)	10,279 (5.6)	10,020 (5.5)	-259 (-3)
Meadow / Grass (MG)	76,962 (42.2)	57,846 (31.7)	-19,116 (-25)
Shrub (SH)	6,490 (3.6)	868 (0.5)	-5,622 (-87)
Woodland (WD)	12,591 (6.9)	7,742 (4.2)	-4,849 (-39)
Broadleaf Deciduous (BD)	7,062 (3.9)	10,251 (5.6)	+3,189 (+45)
Mixedwood (MX)	5,279 (2.9)	9,500 (5.2)	+4,221 (+80)
Conifer (CF)	63,920 (35.0)	86,356 (47.3)	+22,436 (+35)

Table 4.5: Vegetation change on the landscape classified as direction of succession. If vegetation was in a more advanced successional state in 2008 relative to 1909, it was considered “forward”, if the same it was considered “same”, if it was in an earlier succession state it was considered “reverse”. This is shown for the whole landscape, and also broken down by Natural Subregion (Natural Regions Committee 2006). See also Fig. 4.5.

Vegetation Succession Direction	Total Landscape (%)	Natural Subregion				
		Alpine (%)	Subalpine (%)	Montane (%)	Foothills Parkland (%)	Foothills Fescue (%)
Reverse	8.7	4.2	9.3	10.2	14.0	4.4
Same	63.4	70.1	64.7	48.3	56.6	93.3
Forward	27.8	25.7	26.0	41.6	29.4	2.3

Table 4.6: Vegetation class transitions between 1909 to 2008. Each column adds up to 1 to account for all vegetation of that category and what it transitioned to 99 years later (may not all add to 1.0 due to rounding errors). The percentages associated with each vegetation category reflect the proportion of the total landscape occupied by that vegetation type for that time period. Grey shaded cells remain in the same category, cells above this shaded line are in an earlier successional state, and cells below the shaded line are in a more advanced successional state in 2008 relative to 1909. NV= nonvegetated, MG = meadow/grass, SH = shrub, WD = woodland, BD = broadleaf deciduous, MX = mixedwood, CF = conifer.

	Vegetation 1909						
Vegetation 2008	NV (5.6%)	MG (42.2%)	SH (3.6%)	WD (6.9%)	BD (3.9%)	MX (2.9%)	CF (35%)
Reverse	**n/a**	**.016**	**.165**	**.130**	**.237**	**.247**	**.141**
NV (5.5%)	.795	.016	.009	.018	.011	.013	.002
MG (31.7%)	.083	.611	.156	.111	.212	.082	.088
SH (0.5%)	.004	.007	.034	.001	.006	.003	.000
WD(4.2%)	.054	.051	.040	.102	.008	.028	.024
BD (5.6%)	.300	.051	.311	.033	.412	.121	.005
MX (5.2%)	.005	.041	.192	.077	.198	.246	.021
CF (47.3%)	.057	.222	.259	.659	.154	.507	.859
Forward	**.205**	**.372**	**.802**	**.769**	**.352**	**.507**	**n/a**

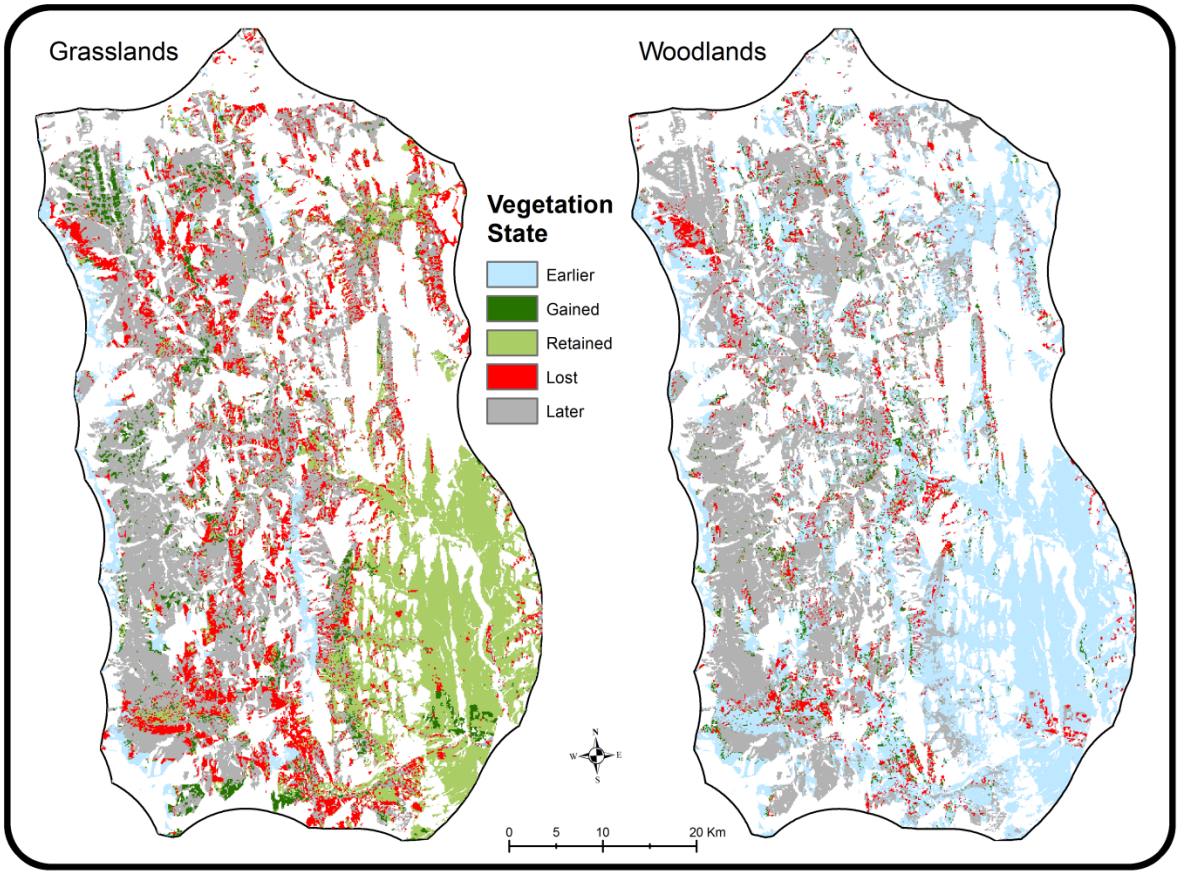

Figure 4.6: Grassland and canopy woodland change since 1909. The vegetation state refers to the condition of targeted vegetation (grasslands and woodlands) categories in 1909 and whether they were gained (not present in 1909, present in 2008), lost (present in 1909, not present in 2008) or retained (present in 1909 and 2008) in 2008. Earlier and Later refer to the surrounding vegetation matrix in both 1909 and 2008: "Earlier" means the vegetation was in a successional state earlier than the target vegetation category (nonvegetated for Grassland map, nonvegetated + grassland + shrubland for Woodlands map). Later means the vegetation was in a more advanced successional state than the target vegetation category (shrub + woodland + mixedwood + broadleaf deciduous + conifer for Grassland map, mixedwood + broadleaf deciduous + conifer for Woodlands map).

4.4.2. ***Correlates of Landscape Change***

While the majority of the landscape was in the same successional state it was in 1909, and a considerable amount was in a more advanced successional state, some of the landscape was in an earlier successional state in 2008 than it was in 1909. For this to occur, some disturbance must have caused this shift. Not all disturbances result in shifts to earlier succession states, but nearly all of the "reverse succession" (87%) was associated with

areas that have undergone harvesting, wildfire, or other anthropogenic disturbance between the two time periods (Table 4.7). For the remaining 13% of the landscape that has undergone reverse succession we were either missing the records (harvesting, fire, disturbance) that would account for the change, or some other disturbance occurred (windthrow, landslides, flooding, land clearing).

Table 4.7: Area of the landscape undergoing "reverse succession" and percent of this that was associated with different types of known disturbance in these areas. See also Tables 4.4, 4.5.

Landscape Portion	Area (ha)	Cumulative % explained
Reverse Succession Total Area	15,942	0
Fires	3,756	26.3
Harvested	4,170	52.5
Anthropogenic disturbance	5,527	87.2
Area Remaining	2,489	12.8 (unexplained)

Total Vegetation Change: We found evidence of spatial autocorrelation in the response variable "successional change" (Moran's I-statistic = 0.4203, z-score = 784.57, $p < 0.000001$). This indicated clustering in successional change, with a distance threshold of 1118 meters. We included the random effect of blockshed in the ordinal logistic regression to control for this spatial autocorrelation. All the predictor variables (solar radiation, elevation, time since harvest (TSH), time since fire (TSF), and anthropogenic disturbances (AD)) were included in the best model for the ordinal logistic regression of the response variable (successional change) (see Table 4.8 and Appendix C for a summary of all model outputs).

Across the entire study area, growing season solar insolation had a mean value of 829,647 WH/m^2 and the 25th and 75th percentile range was 806,157 – 871,165 WH/m^2. Solar insolation was negatively related to forward successional change (higher insolation = less change, lower insolation = more change). With harvesting in the 2000s as the reference condition, sites with a longer time since harvest were more likely to have returned to or surpassed their pre-disturbance successional state. With fires in the 2000s as the reference period, areas that burned earlier than this had an increased probability of forward succession. Elevation was positively related to the probability of forward succession, while the presence of anthropogenic disturbance decreased the probability of forward succession. Spearman rank correlations revealed no strong correlations between the predictor variables

(see Appendix C). Elevation and AD showed a weak correlation (rho) of -0.32, indicating that most anthropogenic disturbance occurred at lower elevations.

Table 4.8: Outputs of the best ordinal logistic regression model which included all predictor variables for the direction of landscape vegetation change. The response variable is the "succession" variable with values of -1 for reverse succession, 0 for no change, and 1 for forward succession (see Fig. 3.5). The reference categories for TSH (time since harvest) and TSF (time since fire) are harvesting in the years from 2000-2008, and fires in 2003-2007 respectively. Z.solar and Z.elev indicate solar insolation and elevation transformed to z-scores. AD is the presence or absence of anthropogenic disturbance. See also Figs. 4.4 and 4.5.

Variable	Estimate	Std. Error	Z-value	Pr (>|z|)
Z.solar	-0.036	0.005	-7.139	< 0.001
TSH (1990s)	0.397	0.043	9.282	< 0.001
TSH (1970-1989)	1.707	0.047	36.330	< 0.001
TSH (1950-1969)	1.610	0.059	27.078	< 0.001
TSH (never)	2.489	0.032	77.622	< 0.001
TSF (1930-1945)	1.842	0.062	29.945	< 0.001
TSF (1910)	0.243	0.060	4.023	< 0.001
TSF (<1910)	2.150	0.058	36.903	< 0.001
AD	-0.069	0.011	-6.068	< 0.001
Z.elev	0.209	0.009	23.351	< 0.001

Meadow and Grassland (MG) and Woodland (WD) Change: We found evidence of spatial autocorrelation in both the MG and WD change response variables (Moran's I-statistics = 0.596 (MG), 0.490 (WD), z-scores = 734.71 (MG), 123.9 (WD), p-values for both < 0.000001). These results indicated clustering in the response variables with distance thresholds of 2561.5 m (MG) and 2524.1 m (WD). We included the random effect of blockshed to control for this spatial autocorrelation. For areas categorized as MG and WD in 1909, three predictor variables (solar radiation, elevation and time since fire) were retained in the best model (Tables 4.8 and 4.9 and see Appendix C for a summary of all model outputs).

In the portion of the landscape occupied by MG in 1909 the mean solar insolation value was 833,640 WH/m^2 with the 25th and 75th percentile range of 818,159 – 858,229 WH/m^2, which is a narrower range than for the whole landscape. In the portion of the landscape occupied by WD in 1909, these values were 839,088 WH/m^2 (mean), 807,257 WH/m^2 (25th percentile) and 885,309 WH/m^2 (75th percentile), which indicates an overall

higher level of insolation than for the whole landscape. For both the MG and WD portions of the landscape in 1909, and consistent with what we saw for all vegetation types combined, sites receiving less solar radiation had an increased probability of forward succession (Tables 4.9 and 4.10).

Table 4.9: Outputs of the best ordinal logistic regression model for the magnitude of change for areas that were meadow or grassland in 1909. The response variable is the magnitude of change variable with values of 0 for no change, 1 for change to shrubland or open canopy woodland, 2 for change to broadleaf deciduous or mixedwood, and 3 for change to conifer (see Table 4.3). The reference category for TSF (time since fire) is fires that burned in 2003-2007. Z.solar and Z.elev indicate solar insolation and elevation transformed to z-scores. See also Figs. 4.4 and 4.5.

Variable	Estimate	Std. Error	Z-value	Pr (>|z|)
Z.solar	-0.545	0.010	-56.49	<0.001
TSF (1930-1945)	1.602	0.100	16.05	<0.001
TSF (<1910)	1.582	0.093	16.92	<0.001
Z.elev	1.074	0.017	63.80	<0.001

Table 4.10: Outputs of the best ordinal logistic regression model for the magnitude of change for areas that were open canopy woodland in 1909. The response variable is the magnitude of change variable with values of -1 for change to grassland or shrubland, 0 for no change, 1 for change to broadleaf deciduous, 2 for change to mixedwood or conifer (see Table 4.3). The reference category for TSF (time since fire) is fires that burned in 2003-2007. Z.solar and Z.elev indicate solar insolation and elevation transformed to z-scores. See also Figs. 4.4 and 4.5.

Variable	Estimate	Std. Error	Z-value	Pr (>|z|)
Z.solar	-0.147	0.022	-6.686	<0.001
TSF (1930-1945)	3.654	0.294	12.440	<0.001
TSF (<1910)	4.320	0.285	15.162	<0.001
Z.elev	-0.146	0.037	-3.991	<0.001

For both vegetation types, areas that burned in 1930-1945 showed a higher probability of forward succession as compared to areas that burned in 2003-2007. For areas that were MG in 1909 there was little difference in the probability of forward succession for areas burned in 1930-1945 and those that burned pre-1910. For areas that were WD in 1909 those that burned pre-1910 showed an even greater probability of forward succession than those that burned between 1930-1945. Elevation had a positive effect on the probability

of forward succession in the MG portion of the landscape but a decreased probability in the WD portion. There were no strong correlations between any of the predictor variables (Appendix C).

We found that more than 94% of the grasslands that converted to forest between 1909 and 2008 occurred adjacent to existing forest stands. Less than 6% of the grasslands lost were the result of new forests growing in the middle of pre-existing grasslands. So too with open canopy woodlands, we found that nearly 85% of the area lost was adjacent to pre-existing forest, with roughly 15% adjacent to other woodlands, or surrounded by grasslands or non-vegetated areas (Table 4.11 and Figure 4.6). Grassland and woodland losses by Natural Subregion are shown in Table 4.12 The Chi-square analysis showed that both grassland and woodland losses were significantly different among the Natural Subregions at a probability value of less than 0.0001 (Pearson's Chi-squared = 27910 (MG) and 3109.7 (WD), degrees of freedom = 8 for both).

Table 4.11: Amount (proportion of area, and length of edge) of patches of 1909 grassland and woodland losses neighboring neighbouring historical forest or non-forest. The "Edge with Neighbor" column shows the total length and proportion of edges that had neighbors, because some woodland and grassland patches bordered non-visible portions of the landscape. Of these edges with neighbours, "Edge Touching Forest" and "Edge Not Touching Forest" measured the percentage of edge length shared with forest or not (for Grassland, this includes open canopy woodlands and all forest types; for Woodland this includes all forest types). "Area Touching Forest" measures precentage of area associated with these patches touching forest. "Area Not Touching Forest" measures the percentage area of patches that were completely isolated from forests.

Historic Vegetation Lost	Area Lost (ha)	Edge with Neighbor (km)	% Edge Touching Forest	% Edge Touching Not Forest	% Area Touching Forest	% Area Not Touching Forest
Grassland	29,936	3930 (73.2%)	67.2	32.6	94.2	5.8
Woodland	11,311	1971 (75.7%)	45.6	54.4	84.5	15.5

Table 4.12: Proportion of area that was meadow/grasslands (MG) and open canopy woodlands (WD) in 1909 that was lost by 2008 (changed to earlier or later successional stages) by Natural Subregion throughout the study area.

Vegetation Succession Direction	Natural Subregion									
	Alpine (%)		Subalpine (%)		Montane (%)		Foothills Parkland (%)		Foothills Fescue (%)	
	MG	WD	MG	WD	MG	WD	MG	WD	MG	WD
Reverse	9.8	10.3	1.7	8.1	1.8	12.1	0	20	0.1	97.8
Same	35.7	13.8	19.6	11.3	58.2	8.9	73.4	0	97.0	0.7
Forward	54.5	75.9	78.7	80.5	40.0	79.0	26.6	80	2.9	1.5

4.5. Discussion

4.5.1. *Vegetation Change 1909-2008*

By using a large number of repeat photographs (137 image pairs) covering such a large landscape (~320,000 ha) to spatially quantify vegetation change since the time of European settlement, we have been able to gain new insights into landscape change over this time period. In terms of the overall successional state of the landscape, we saw that the great majority of the landscape has remained in the same vegetation category from 1909-2008; however, more than a quarter (28%) of the landscape is in a later seral stage than it was in 1909, while less than 9% is in an earlier state. The landscape has shifted from a condition in 1909 with nearly 58% in a non-forest state (grasslands, meadows, shrubs, non-vegetated and open canopy woodlands) and 42% in closed canopy forest (conifer, broadleaf deciduous and mixedwood) to 2008 where 42% was in non-forest and 58% in forest. While there was some movement in the other direction (conversion of forests to non-forest), the net balance has clearly shifted in favour of closed canopy forests.

Johnson and Fryer (1987) examined vegetation change in the Kananaskis Valley roughly 80km to the north of our study area and concluded that there was no evidence that European settlement and fire suppression had caused any landscape scale vegetation changes. They used forestry surveys conducted in 1883 and 1972 and found that sites occupied by lodgepole pine and Engelmann spruce remained largely the same over this time period. By focusing solely on sites with commercially valuable trees (only conifer) in 1883, their analysis did not include any portions of the landscape that might have been more

open canopy woodland or grasslands; thus, they could not have detected any conversion of open woodlands or grasslands to closed canopy forest, or conversion of aspen to conifer.

We could not reliably differentiate age- or size- classes of conifer trees in the original black and white photography, and therefore had to treat all conifer forest as the same class. Due to this limitation we cannot say whether the conifer forest itself changed or not between 1909 and 2008. Had we been able to factor age-class distributions into our classification of vegetation by differentiating between immature, mature and overmature forests, we likely would have seen a shift in the CF class towards older age classes between 1909-2008 and would have found less area that had not changed. Johnson et al. (1994) examined age-class distributions within *Pinus contorta* and *Picea engelmanii* forests in the same geographic area where Johnson and Fryer (1987) concluded that the species composition of the forest had not changed, and showed that the age class distribution of the forest had changed markedly with predominantly younger trees in the 1800s to older trees by the 1980s. These shifts in age class distribution in coniferous forests over this time period have also been shown throughout the Rocky Mountains by Gruell (1983), Andison (1998), and Rhemtulla et al. (2002).

4.5.2. ***Factors Contributing to Spatial Variability in Vegetation Changes***

4.5.2.3. Natural Subregions

The changes in vegetation class were not homogenous across the entire landscape. We found the Montane Natural Subregion had the greatest proportion of area undergoing successional advancement, with substantial (but lesser) forward change in the Subalpine, Alpine and Foothills Parkland. With more closed canopy coniferous forest in the Subalpine Natural Subregion, part of the reason it did not change as much as the Montane was that more of the landscape was already at the endpoint of our successional sequence (CF). Conversely, in the Montane, there was substantial cover of early successional stages in 1909, and therefore a much greater proportion of the landscape could succeed. The vegetation classes within the Foothills Fescue Natural Subregion remained largely unchanged, but as with forest structure, we did not differentiate grasslands or meadows in any way other than to define them as areas with no significant amount of trees or shrubs on site. The majority of the forward successional change in the Alpine Natural Subregion is likely due to the advancement of treeline upslope; however, we did not measure this explicitly (also discussed below).

With regard to grassland and open canopy woodland loss alone, 13% of the total landscape that was historical grasslands had become closed canopy forest and 5% of the landscape that had been open canopy woodland had converted to closed canopy conifer forest by 2008. Of the original grasslands present 1909, just over 37% had converted to a more successionally advanced vegetation type in 2008. These losses in grasslands were partially offset by a gain of nearly 10,000 ha due to disturbances within other vegetation types, resulting in a net loss of 25% in grassland area between 1909-2008. Similarly for open canopy woodlands, 77% of the open canopy woodlands from 1909 succeeded to more advanced vegetation types in 2008, however due to nearly 6,500 ha of other vegetation types converting to open canopy woodlands, there was only a net loss of 39% in area between 1909-2008. These new open canopy woodlands might represent advancing forest edges (seeding in front of the main forest margin) or disturbances breaking up closed canopy forests. We saw that nearly 80% of open canopy woodlands converted to more advanced successional types in the Alpine, Subalpine, Montane and Foothills Parkland Natural Subregions, and this appears to be due to gradual forest infill given that the majority of the area lost in this vegetation type was adjacent to existing higher-density forests. There was more variability among Natural Subregions in the proportion of grasslands lost, with much higher losses in the Subalpine Natural Subregion compared to the Montane and Alpine. As with open canopy woodlands, most of the grasslands that converted to more advanced successional types appear to be the result of forest advancement from the historic forest edge given that nearly all of this “new forest” is directly adjacent to the old forest edge. The grassland losses in the Montane were considerably lower than they were in the Subalpine, perhaps due to less forest edge available from which the forest could encroach. In the Alpine, the grasslands and meadows can only be lost due to forest advancement from one side, as the other side borders on bare rock and talus slopes.

Montane vegetation in general tends to be under strong edaphic control (Archer 1994; Natural Regions Committee 2006), and visual inspection of the historic and repeat images used in this study showed very clear patterns of interspersed grassland and forest driven by aspect throughout the Montane region of our study area, and along the valley bottoms and higher elevation ridges (treeline) in the Subalpine and Alpine. Two possible mechanisms that might describe the increase in tree cover are either the climate regime has changed between 1909-2008 to create conditions more favourable for tree establishment, or that historic disturbances regimes (fire, grazing) kept these areas in a grassland state in

1909, but the frequency or severity of disturbance has since been reduced allowing for greater tree recruitment. These will be discussed further below.

Other data sources also confirm our observations of forest encroachment into grasslands. Strong (1977) used pollen from sediment cores extracted from lakes throughout southern Alberta to compare pre- and post-settlement (1800s – 1900s) vegetation, and found that what was considered "aspen parkland proper" in the 1970s used to be "groveland", and what was "groveland" in 1977 had previously been fescue grasslands. Campbell et al. (1994) showed a biome wide expansion of aspen into former grasslands dating to the 1880s and 1890s along the margin of what is now referred to as the aspen parkland all the way from Regina, through to Edmonton and south to Calgary, which is just north of our study area. Within our study area, an unpublished study by the Alberta Conservation Association (Didkowsky and Albricht, 2009) showed changes in the Bob Creek Wildland grassland cover between 1950-2009 using aerial orthophotos. They found that the loss of grasslands was occurring at a rate of roughly 5 ha/year in the Montane portion of the Bob Creek area (very roughly 4-5% of the total grasslands over 50 years). They also found that roughly 10% of grasslands were lost between 1950-2009 in the Porcupine Hills, immediately to the east of our study area. Neither of these rates is far outside our observations of 37% gross and 25% net grassland loss over a much larger landscape (and considering we examined nearly double the time frame). In the Montane Natural Subregion we found a 40% reduction in grasslands, which is similar to what was observed by Rhemtulla et al. (2002).

4.5.2.4. Topographic effects

We saw an overall positive relationship between elevation and the probability of forward succession of all vegetation types considered together across the entire landscape, regardless of what its vegetation class was in 1909. At high elevations we saw grasslands being converted to forest between 1909-2008 and this is consistent with other studies showing treeline advancement throughout the Rocky Mountains in the 20th century (Luckman and Kavanagh 2000; Klasner and Fagre 2002; Elliott 2011). At lower elevations we saw greater amounts of reverse succession occurring, and this likely relates to higher levels of anthropogenic disturbance, which is concentrated around areas under agricultural management, and tree-removing disturbances within communities and along linear corridors such as roads, powerlines, pipelines, timber harvesting access trails, and railways.

When we examined the degree of vegetation change in historic open woodlands since 1909, we saw that the probability of forward vegetation succession was lower at high elevation than it is at low elevation. At low elevations, 80% of the open canopy woodlands in the Montane and Subalpine have become closed canopy forest. While we also saw large losses of open canopy woodlands to more advanced successional types in the Alpine, many high elevation open canopy woodlands occurred on rocky outcrops, high-slope talus fields, and scattered krummholz stands which could not succeed to closed canopy forest. This might have contributed to the overall negative effect of elevation on forward vegetation succession in former open canopy woodlands.

Insolation was negatively related to the probability of forward succession for the whole landscape, and for historic grasslands and woodlands. For the whole landscape the majority of north and northeast facing aspects (low insolation values) are already treed, and thus unable to move forward in succession. This might be that areas with higher insolation had a lower probability of forward succession as these are the south and southwest facing aspects with moderate slopes. Many of these areas may remain in a grassland state due to physical factors such as lower moisture and higher temperatures. In grasslands, areas with lower insolation values are gentler slopes with north and northeast aspects, and tree germination could occur more easily on these cooler and moister aspects of the landscape. With regard to areas that were formerly open canopy woodlands, we see that the interquartile range in insolation ranges was broader than it was for the greater landscape. Higher insolation values occur on steeper south and southwest slopes. These locations have higher moisture stress, which might limit the potential for vegetation succession. In areas with lower insolation values (cooler and moister aspects), there is a greater probability of tree germination and growth.

4.5.2.5. Disturbance history effects

We saw a net positive relationship between the time since harvest and the probability of forward succession. The most recent harvests had the highest probability of setting the successional sequence back, and all of the areas that had been harvested had forest cover in 1909, so there is no potential for these areas to succeed beyond the state they were in initially. Numerous areas harvested in the 1990s and 2000s still appeared unforested in 2008, however that may well be because small seedlings would not have been visible in the

photographs, and also, some of these harvested areas that were harvested were salvage logged following the Lost Creek fire in 2003.

As with the time since harvest, recent fires have had a significant effect on the successional state of the landscape, relative to what the vegetation was before the disturbance by setting the successional sequence back. In 2003, the Lost Creek Fire burned 6,100 ha in our study area (although only 2,500 ha in our visible landscape) and only 25% the burned area was grassland in 1909, whereas 50% was classified as grassland in 2008. As the time since fire lengthened, there was a higher chance of vegetation recovering its predisturbance state or even going past it in the successional sequence. The 1910 fire was defined by burned forest, so these areas could not go past what they once were, which likely explains why it had the lowest estimate for change. Pre-1910 fires could occur in any vegetation type, as could 1930s-1940s and 2000s fires, and so there were more opportunities for vegetation to become more advanced in the successional sequence than they were at the time of the disturbance.

The vast majority of anthropogenic disturbance occurred in lower elevation areas that had been grassland, shrubland and open canopy woodland in 1909, therefore the potential for vegetation to be moved backward in the successional sequence was somewhat diminished for the whole landscape.

4.5.3. *Fire, Climate, Grazing, and Vegetation*

We saw an overall seral stage advancement of much of the landscape, which suggests changes in the disturbance regimes of the landscape. The two primary disturbance agents operating in the past centuries on this part of the landscape were wildfire and large animal grazing. In the 1913 imagery we found ~14,000 ha burned by the 1910 fire (nearly 8% of the visible landscape), and saw evidence of numerous other relatively recent fires across large areas of the landscape. These other fires likely burned in the 1880s and 1890s based on the height of the regenerating trees, and based on the vast areas of large downed woody debris that were still in very early stages of decomposition. The stage of decomposition was apparent due to branches and twigs still intact and visible in the photographs. Rogeau (2016) summarized the findings of many fire regime studies throughout the southern Alberta Rockies and found historic (pre-1900) mean fire return intervals in the open Montane Natural Subregion areas (outside the mountains) were less than 50 years, and in the more enclosed Montane regions (low elevation valleys running into

the mountains surrounded by Subalpine areas) were less than 145 years. In the Subalpine Natural Subregion, the mean fire return interval in the area between the Highwood Pass and Waterton Lakes National Park (containing our study area) was 36-62 years (Rogeau 2016). While mean fire return intervals do not equate directly with fire cycles, if we adopt a very conservative estimate of the fire cycle being 100 years in the Montane and Subalpine, we should expect to see an average of 1% of the landscape burn per year. While we have not measured fire return intervals or fire cycles, our general observations of area burned in the late 1800s and early 1900s are consistent with studies of historical fire regimes in the region (Tande 1979; Hawkes 1979; Barrett 1996; Rogeau 1999; Rogeau 2004; Rogeau 2005a; Rogeau 2005b; Rogeau 2008). Since 1913, wildfire has burned a total of only 8.5% of our visible study area over 113 years (since 1913), which is a mean annual area burned of 0.075% (a fire cycle of 1,333 years). Tymstra et al. (2005) showed that for the entire Montane and Subalpine Natural Subregions in Alberta the mean annual area burned was less than 0.02% between 1961-2002.

Numerous studies describe the effects of 20th century fire exclusion on vegetation change (Baker 1992; Arno et al. 2000; Gallant et al. 2003; Daniels et al. 2011). Vegetation succession can occur when fire is removed if it was historically limited by regular fire activity. The probability of fire is not uniform across the landscape, and is affected by fuels, weather, and topography. Cooler slopes (lower insolation) on north and northeast aspects will burn less than warmer and drier slopes (high insolation) on south and southwest aspects. However, in the right climatic conditions, cooler slopes can be vulnerable to extreme fire behaviour resulting in very large areas burned (Stocks et al. 2002; Taylor and Skinner 2003). Historical variation in fire frequency and intensity correlates strongly with climatic variability (Brown 2006; Gedalof 2011). During moderate, or “normal” climate periods, there would be more frequent fire of variable intensity on the warmer and drier parts of the landscape (Rogeau 2016). When climate conditions are more extreme (dry, hot, windy), there would be less frequent and higher intensity fire on the cooler and moister parts of the landscape, as fires burn more indiscriminately across the landscape, regardless of fuel type in these conditions (Miller and Urban 2000). It is difficult to prove that changes in fire regimes are directly responsible for the widespread losses of open grasslands and open canopy woodlands because some of the same variables that drive variability in fire behaviour (elevation and insolation) affect variability in vegetation, and it is difficult to tease apart these confounding effects. However, it is also difficult to imagine that with the high success rate of wildfire suppression since the early- to mid-1900s that we have not had an

influence on the disturbance regime and hence vegetation succession patterns of the landscape (Cumming 2005; Pyne 2011). Furthermore, given the complex feedbacks between climate, vegetation, and disturbance regimes (Flatley and Fulé 2016), there is the potential for the creation of novel ecosystems in both the near and long term future (Hobbs et al. 2014).

Variability in climate is another factor that cannot be ignored. Edwards et al. (2008) used tree rings to reconstruct the climate over the past 1000 years in the Alberta Rocky Mountains and found that: a) growing season relative humidity from 1700-1900 was considerably lower than from 1900-2000; b) there was a pronounced drought from 1900 to ~1930, followed by a significant wet period beginning in the 1930s; and c) it has been considerably warmer from 1901-1990 than it was from 1531-1890. Luckman et al. (1997) showed that during the 1930s there were several years with summer temperatures 1.0 - 1.5 ^{o}C above the 1961-1990 mean (coinciding with the wet period described by Edwards et al. (2008)), while most of the late 1800s and earlier 1900s were 1.5 - 2.0 ^{o}C below this mean. In the absence of disturbance, the climatic conditions in the 1930s could well have contributed to a large pulse of tree recruitment that had previously been limited by the 1900-1930 drought. To confirm this, we would need to study the age class distribution in new forests occupying former grasslands and see when these forests began to grow.

Succession of grasslands to forests at lower elevations may also have been influenced by changes in other disturbances such as grazing (Campbell et al. 1994). In this region there were very large herds of bison and elk in the grasslands and Montane portions of the landscape (Brink 2008), both of which would have browsed aspen and grasses extensively and had a large effect on the balance between grasslands and forest (Bachelet et al. 2000; White 2001). Bison herds were extirpated in the 1880s, and Campbell et al. (1994) showed a large expansion of aspen that coincides with the removal of bison. While we could not tell the age of the stands of aspen throughout the Montane and foothills fescue from the historical photographs, there were large areas of young aspen stands that may well date to the time of the bison extirpation matching what Campbell et al. (1994) observed over an even wider geographic area.

With only two points in time, we know neither the rate nor the dynamics of how these former grasslands and open canopy woodlands transitioned to forest over the 100 year period. If the movement of the forest-grassland ecotone in favour of trees has occurred as a steady advancement from the historic forest edge, we would expect to see a decrease in tree ages with increasing distance from these edges. If we saw trees of the same age

regardless of distance from the historic edge (indicating that large areas became treed in a single pulse), it might indicate that while there was a seedbed available, the climate was previously unfavourable for seedling growth or seedlings were able to germinate but disturbance prevented them from growing (fire or heavy grazing). Once the climate shifted, or the disturbance was removed (or both), the seedlings were able to germinate and grow.

Schneider (2013) modeled future climate change throughout Alberta and found that over the next 80 years in our study area the climatic envelopes most suitable to vegetation in the Foothills Fescue will expand into areas that are currently classified as the Montane Natural Subregion, which in turn will expand into what is currently the Subalpine, which in turn will encroach upon the Alpine. As a result of these forecast changes, we should be expecting to see grassland/forest ecotone shift in favour of grassland at lower elevations. Meanwhile, what we have witnessed over the past 100 years is vegetation change in the opposite direction, with trees moving into these grasslands. We have lost much of the open canopy woodlands that typify the Montane Natural Subregion, and in turn we see closed canopy forests that are more typical in the Subalpine Natural Subregion beginning to occupy and dominate in what is currently classified as the Montane. We are also seeing the vegetation that largely defines the Montane Natural Subregion moving into the grasslands. We can think of this as increased ecological tension in the system as the vegetation on a particular site becomes further removed from that which is best suited for the climate of that site.

Does this mean that we should consider restoring historical vegetation on the landscape today to mitigate this ecological tension? To view ecological tension only in terms of a disconnect between climate and vegetation ignores the potential for tension in the three-way interaction of climate, vegetation, and disturbance regimes. Changes in climate likely have, and will continue to affect fire regimes into the future as climate governs the weather conditions under which fires burn, and alters the frequency and location of lightning ignitions. Similar to our findings, but on the rim of the Grand Canyon, Flatley and Fulé (2016) also found that historical vegetation structure (from the 1900s) was better suited to projected climate change than the current vegetation. However, when they examined how fire regimes would be altered in the future due to the projected changes in climate, they found that the altered fire regime would not support the historical vegetation structure. They concluded that this would likely result in novel ecosystems emerging in the future (Flatley and Fulé 2016). Hobbs et al. (2014) argued that interventionist management in highly altered ecological systems needs to move beyond simplistic restoration approaches and

consider and value the potential of novel ecosystems.

4.5.4. ***Implications and Conclusion***

While our findings showed a great deal of change in vegetation structure on the landscape that is consistent with other studies of change over this time period, it also showed the majority of the landscape was in the same broad vegetation category in 2008 as it was in 1909. However, to say that most of the landscape has not changed would misrepresent our findings, and we need to consider the following caveats: A) the closed canopy coniferous forest category did not account for changes in species composition or age class structure; b) the grassland and meadow category did not differentiate between true grasslands and heavily grazed agricultural land, cropland, and human maintained clearings around settlements and infrastructure; c) the mixedwood category encompassed considerable variation in the ratio of broadleaf deciduous to coniferous trees; d) the open canopy woodland category varied from grasslands intermingled with single or clustered large *Pseudotsuga menziesii* trees to talus slopes with krummholz to areas with mostly grassland with interspersed aspen copses.

With these broad categories we were able to show a large loss of grasslands and open canopy woodlands throughout this landscape. We are seeing an increased ecological tension between the direction of vegetation change occurring presently in favour of forest expansion, and the influence of future climate change which should instead be driving these vegetation changes in favour of grassland expansion. The causes of these changes in vegetation structure are complex, and these changes are influenced by topography and disturbance history either directly or indirectly through other mechanisms that correlate to these factors.

Some find this forest expansion into grasslands to be cause for ecological concern (Arno and Gruell 1983; Archer 1994; Noss 2013), while others might feel that we are just in an interregnum in wildfire activity that will eventually “catch up” (Weir et al. 1995) and restore the grassland-forest boundary to where it is most climatically suitable. We do indeed have ample proof that massive wildfires have burned at high severity through this landscape in the past (1910, 2003) and could do so again in the future. When the climatic conditions are suitable, large wildfires could burn back the forests that have encroached into grasslands

and if the climate is indeed unsuitable for tree germination and growth, these areas will remain as grassland so long as it is the most suitable vegetation for the climate.

While some people may not support direct intervention with management, others advocate for more active management of the landscape to ensure “ecological integrity”, especially in protected areas (Jackson and Hobbs 2009). Agencies in both Canada and the USA have active programs designed to assess changes in fire regimes, and address the ecological consequences of these changes (White et al. 2003; Barrett et al. 2010). Within our study area there are numerous protected areas (Castle Crown Wilderness Area, Bob Creek Wildland, Black Creek Heritage Rangeland, Beehive Wilderness Area, Mt Livingstone Natural Area, Plateau Mountain Ecological Reserve, and portions of the Don Getty Wildland). The management plan for the Bob Creek Wildland and Black Creek Heritage Rangeland explicitly states that fire suppression and exclusion have had detrimental effects on their ecological integrity, and that some restoration to historical conditions is highly desired (Government of Alberta 2011). This study can help to determine what ecological conditions were like at the turn of the 20th century, as compared to currently, and this will help guide restoration efforts in some areas of the landscape. Enacting a historical landscape restoration plan across a large area would represent a large-scale experiment in landscape diversification, however, any such plan must consider the potential emergence of novel ecosystems.

Many land managers and scientists question why we should look to the past to guide our management actions in the present under a changing climate that will likely result in novel conditions and ecosystems (Klenk et al. 2008; Kramkowski 2012). Swetnam et al. (1999), Bjorkman and Vellend (2010), Higgs et al. (2014), and Stockdale et al. (2016) all make compelling arguments that understanding reference conditions at key moments in ecological history can help managers to better project the future trajectory of current systems under different scenarios. By no means does “ecological restoration” mean perfectly recreating historic conditions to ensure that ecosystems maintain their resilience in the future, but we do need to understand ecological history.

This study applied new technology (Bozzini et al. 2012) and novel techniques (Stockdale et al. 2015) in oblique image analysis to extend the temporal window of our understanding of landscape vegetation change to the beginning of European settlers’ arrival in southwestern Alberta. While much of this is a descriptive study, we also tested several hypotheses to examine the relationships between vegetation change, topography and disturbance history. From these relationships we intend to continue testing and refining

hypotheses in future studies to better understand the mechanisms behind the observed variation in change by focusing our investigations on key areas of vegetation transition. Future work is already planned to refine analysis methods to permit finer scale historical image analysis at a pixel scale which will enable us to investigate more closely the mechanisms responsible for grassland and woodland transitions to closed canopy forests, and to examine whether or not the forest itself has changed with regard to species, density, and size class distributions.

Chapter 5: Using historic landscape vegetation structure for ecological restoration: effects on burn probability in the Bob Creek Wildland, Alberta, Canada.

5.1. Abstract

In montane regions throughout western North America we have seen large scale forest encroachment on grasslands, which many people suspect is the result of climate change and lengthening of fire return intervals due to fire suppression and exclusion. The conversion of grasslands and open canopy forests to closed canopy coniferous forests threatens many ecological values in montane regions, and brings with it a concomitant increase in the probability of high intensity wildfire. Many agencies are building plans for the purpose of, and actively restoring landscapes to their historical condition under the assumptions that this will reduce the probability of high intensity wildfire and preserve the ecological integrity of the landscape. We used the Bob Creek Wildland (BCW) and surrounding landscape in the southern Rocky Mountains in Alberta, Canada, to test whether restoration to the historical landscape condition would: a) reduce the raw burn probability (likelihood of fire at any intensity); b) reduce the conditional burn probability of fire at an intensity greater than 4,000 kW/m (a threshold at which ground attack crews can no longer directly engage with a fire and tactics must switch to aerial attack); c) change the spatial pattern of burn probabilities (raw and conditional) on the landscape; and d) change the distribution of fire sizes on the landscape. We used a subset of the historical photographs from the Mountain Legacy Project (MLP) used in Chapter 4 to reconstruct the vegetation composition from 1909 (historical restoration scenario) and compared this to the current vegetation composition of the landscape (baseline scenario), which was derived from the Government of Alberta provincial fuel grid. Only 58% of the landscape was visible in the MLP photographs from which to determine the historical vegetation composition, and we used Indicator Kriging to interpolate vegetation categories on the non-visible portions of the historic landscape. The historical restoration scenario involved changing the vegetation composition of the BCW to what it looked like in 1909 while leaving the surrounding landscape in its current (2014) condition. We used the Burn-P3 model to calculate the raw and conditional burn probabilities, and the fire size distributions in both scenarios. More than 50% of historic grasslands in the study area have been lost to forest encroachment, and much of what is coniferous forest today used to be broadleaf deciduous and mixedwood forest 100 years ago. The Burn-P3 modelling exercise revealed that the mean raw burn

probability of all fires in the BCW was only very marginally reduced by changing the vegetation (1.3% reduction), and this was even smaller when including a 5km buffer zone around the BCW which had not been restored (0.3% reduction). However, the overall spatial pattern of raw burn probability changed with some areas having burn probabilities in the historical restoration scenario that were half or less of what they were in the baseline scenario, while other areas had burn probabilities 2-5 times higher. When we considered only high intensity wildfires (greater than 4,000 kW/m intensity), we found that the mean burn probability of the landscape was reduced by nearly half (44.2% reduction) in the historical restoration scenario, and many areas had burn probabilities that were 20% or less than they were in the baseline scenario. In areas where the fuel changes resulted in accelerating the rate of spread of fires in the historical restoration scenario we found that the raw burn probability rose, and in areas where the fuel changes resulted in lower rate of spread the probabilities were lower. Furthermore, we found that the mean annual area burned under the current fire environment (considering weather and suppression capabilities over the past 50 years) is expected to be only 3.66 ha using the baseline (2014 vegetation), and 3.71 ha using the historic restoration (1909 vegetation). These translate to fire cycles in excess of 5,600 years, which when compared to the historic fire cycle of 15-30 years in montane areas of the southern Alberta Rocky Mountains, suggest that the modern fire regime is indeed well outside historical norms.

5.2. Introduction

Many studies have shown that forest cover through much of western North America is more homogenous and continuous in the early 2000s than it was at the turn of the 20th century (Arno and Gruell 1983; Gruell 1983; Rhemtulla et al. 2002; Hessburg et al. 2005; Chapter 4). We are only beginning to understand how much the vegetation itself has changed on the landscape; for example, how much grassland and open canopy woodlands have been lost to forest encroachment. Studies have shown that between roughly 1900 – 2000 AD there have been significant shifts from grasslands and open canopy woodlands to closed canopy forests across the forest-grassland interface of the prairie regions (Strong 1977; Campbell et al. 1994), Rocky Mountains (Gruell 1983; Brown et al. 1999; Rhemtulla et al. 2002), and intermountain west in the USA and Canada (Hessburg et al. 2005). We showed (Chapter 4) that over this time period in the southern Alberta Rocky Mountains that 25% and 39% of the grasslands and open canopy woodlands, respectively, have since

converted to later successional stages, while the amount of coniferous-, broadleaf deciduous-, and mixedwood forests have all increased (by 35%, 45%, and 80% respectively). These changes were most pronounced in the Montane and Subalpine Natural Subregions. While we do not know exactly what has caused these increases in forest cover at the expense of grasslands, it is widely believed that 20th century fire suppression and exclusion is one of several key factors (Nelson and England 1971; Arno and Gruell 1983; Archer 1994; Wakimoto and Willard 2005).

It is not inherently problematic that vegetation change is occurring across large areas of the landscape, as all ecosystems have natural ranges of variability (NRV) in species composition and vegetation patterns (Landres et al. 1999). However, considerable evidence suggests that this shift away from open canopy forests, grasslands and meadows is outside of the NRV (Fulé et al. 2002; Agee 2003; Hessburg and Povak 2015), and these changes in vegetation structure across such broad areas have raised concerns regarding ecological values and processes on the landscape. Encroaching forests are a threat to range resources (Gruell 1983; Archer 1994), threaten biodiversity of grasslands at lower elevations (Haugo and Halpern 2007), and of subalpine and alpine meadows at higher elevations (Franklin et al. 1971). Numerous land management agencies have noted the negative effects of losses of grasslands and open canopy woodlands, which are thought to be associated with changes in fire regimes, and have developed management plans to address these (White et al. 2003; Walkinshaw 2008; Barrett et al. 2010; Government of Alberta 2011; Hessburg et al. 2013).

Fire regimes describe how fires interact with ecosystems through space and time (Morgan et al. 2001) and are driven by climate, vegetation, and ignitions (Moritz et al. 2011). Fire regimes are characterized by the cause of ignition (lightning versus anthropogenic), frequency (number of fires per unit time), timing (season of burning), extent (how large individual fires are), and magnitude (the intensity of fires and the severity of their long-term effects). Twentieth century fire suppression efforts have likely altered fire regimes by reducing overall fire frequency and have led to more contiguous coniferous forests that are vulnerable to large crown fires (Baker 1992; Arno et al. 2000; Gallant et al. 2003; Stephens and Ruth 2005). This creates a feedback loop, because by changing the vegetation from grasslands to forests, and from open-canopy and broadleaf deciduous trees to closed canopy conifers, the extent of fires and their magnitude will be changed (Baker 1992; Arno et al. 2000; Marcoux et al. 2015). If the changed extent and magnitude of fires are outside of historical norms, there is a very real risk of ecological damage (Arno et al. 2000). These

changes in fire regimes are further exacerbated by anthropogenic climate change (Carcaillet et al. 2001; Brown et al. 2006). We have seen increases in the overall length of fire seasons in Alberta (Albert-Green et al. 2013) by 46-62 days from 1961 to 2013. The changes in fire season length are also affecting the timing (seasonality) of fire regimes by having more fires burning in the spring in the western USA (Westerling et al. 2006). Many fire ecologists argue that we have already seen significant changes in fire regimes. Studies throughout the Alberta Rocky Mountains (Tande 1979; Hawkes 1980; Barrett 1996; Andison 1998; Rogeau 2005b; Rogeau 2009), British Columbia (Gray 2003; Kubian 2013), and the western United States (Arno 1980; Barrett et al. 1997; Hessburg et al. 2005; Prichard et al. 2009) all show overall declines in annual area burned and lengthened fire return intervals from prior to European settlement in the late 1800s to the present day. These changes coincide with observed shifts in vegetation towards later successional stages (Arno and Gruell 1983; Arno et al. 2000; Hessburg et al. 2013).

Due to these observed changes in vegetation structure and fire regimes, numerous jurisdictions throughout Canada and the USA are investing heavily in thinning forests, changing silvicultural practices, and creating landscape scale ecosystem management plans with the intent of restoring forest age class distributions, species composition and landscape patterns to historic conditions (White et al. 2003; Brown et al. 2004; Baker et al. 2007; Hessburg et al. 2013). The pre-European settlement period is frequently used as a reference point for ecological restoration targets in forest management (Brown et al. 2004; Baker et al. 2007; Barrett et al. 2010; Churchill et al. 2013), rangeland management (Fuhlendorf and Engle 2001) and protected areas management (White et al. 2003; Mawdsley et al. 2009; Bjorkman and Velland 2010; Higgs et al. 2014). Using this reference point as a baseline for determining natural fire regimes, the Nature Conservancy developed the Fire Regime Condition Class assessment tool, which compares the current fire regime and vegetation on a particular site to its historic “norm” and provides a measure of deviation or departure from the NRV (Barrett et al. 2010). Using the FRCC assessment tool, the United States Department of Agriculture, Forest Service (USDA Forest Service) has determined that 26M-33M ha of the ~75M ha of National Forests are in need of restoration to reduce the risk of catastrophic wildfire and/or to bring them within their Natural Range of Variability (NRV) (USDA 2012). In Washington and Oregon nearly 40% of coniferous forests are no longer considered to be within their natural range of variability (Haugo et al. 2015), and in western Montana and northern Idaho (National Forests Region 1) this number is roughly 60% (USDA 2005).

The concept of restoring historical landscapes in the present day is not without controversy, especially in light of anthropogenic climate change that will likely bring novel conditions in the future to which historic ecosystems may be ill-adapted (Klenk et al. 2008). While novel ecosystems will undoubtedly emerge due to climate change (Higgs et al. 2014) it would be misleading to suggest that all historic ecosystems are ill-adapted to future climate conditions (Jackson and Hobbs 2009); they just might require assisted migration in order to ensure the right vegetation is growing under the right climatic conditions (Gray et al. 2011). According to climate projections, we should be seeing the ecotone between forests and grasslands shifting in favour of grasslands in the coming century (Wang et al. 2012; Schneider 2013); however, the grassland-forest ecotone is moving in the opposite direction in favour of forest expansion into grasslands (Chapter 4). As the forest-grassland interface continues to move in favour of forests instead of grasslands, the ecological tension in this system is increasing. Over time the open canopy woodlands of the Montane Natural Subregion will likely disappear, and the flora and fauna that are adapted to grasslands and open canopy woodlands will slowly lose habitat, and species associated with closed canopy lodgepole pine, Douglas-fir, and Engelmann spruce will increase in relative abundance. Some of the underlying assumptions behind management plans designed to change fire regimes and vegetation back to within the NRV is that restoration to historic vegetation composition would improve the ecological integrity of the landscape and reduce the risk of catastrophic wildfires (Shinneman et al. 2012), but this may not hold true across all ecosystems.

This study was designed to test the effects of restoring historical vegetation conditions on wildfire risk using the Bob Creek Wildland in southern Alberta as a case study. This research project and the study area are both a smaller piece of the Land Use Framework planning by the Government of Alberta (Government of Alberta 2014) for the entire South Saskatchewan Regional Plan (SSRP) area. I tested the following expectation:

1. The fuel composition of the Bob Creek Wildland has changed from the time of European settlement to today due to forest encroachment on grasslands, canopy closure in open woodlands, and conversion of broadleaf deciduous forests to conifer.
2. If we could restore the landscape to something similar to the vegetation composition of the historic landscape, and compare it to the conditions of today (as of 2014), it would affect the wildfire regime as follows:

a. Change the burn probability of all wildfires due to these changes in fuel composition. This change may result in increases or decreases in the burn probability.
b. Reduce the burn probability of fires over 4,000 KW/m (crown fires) due to the changes in fuel.
c. Result in changes in the distribution of fire sizes on the landscape with a greater relative frequency of small versus large fires.

This study compared two scenarios: 1) the baseline, which is the vegetation composition of the Bob Creek Wildland and surrounding area as of 2014; 2) the historical restoration, which is the historical vegetation of the Bob Creek Wildland as it was in 1909 embedded in the matrix of the current landscape as of 2014.

5.3. Methods

5.3.1. *Study Area*

The Bob Creek Wildland (BCW) is a 20,775 ha Provincial Wildland Park (Figure 5.1) located in southern Alberta, with the Porcupine Hills immediately to the east, and the Livingstone Range along its western edge. It ranges in elevation from 1345 m – 2210 m above sea level, and is comprised of the Subalpine (8,511 ha) and Montane Natural Subregions (12,264 ha) (Natural Regions Committee 2006). Soils are predominantly chernozems in the Montane, and brunisols in the Subalpine areas. Long north-south rocky ridges over geological faults give the region its characteristic exposed rocky ridges known as the Whaleback. Numerous creeks flow out of the Wildland, however, there are no large standing bodies of water in the area. The study area is at the edge of the Cordilleran and Grassland ecoclimatic provinces. The Cordilleran ecoclimatic province has cold winters, very short cool summers, and precipitation varies due to aspect and elevation (Natural Regions Committee 2006). There is prevailing westerly airflow, however this varies considerably by topography (Natural Regions Committee 2006). The Grasslands ecoclimatic province has cold winters and short hot summers, with most rain falling in June, but overall low precipitation (Natural Subregions Committee, 2006). Strong westerly winds can occur here (Natural Subregions Committee 2006). The mean precipitation and temperature by month during the fire season as recorded at the closest weather station (Livingstone Gap Fire Lookout Tower) from 1983 – 2016 are: April, 27mm, 6.5°C; May, 64mm, 11.7°C; June, 107mm, 15.5°C; July, 51mm, 20.2°C; August, 51mm, 19.8°C; September, 45mm, 15.7°C;

and October, 29mm, 8.3°C. The mean frost-free period between 1961-2010 (Alberta Agriculture and Forestry 2016) was June 9 to August 30.

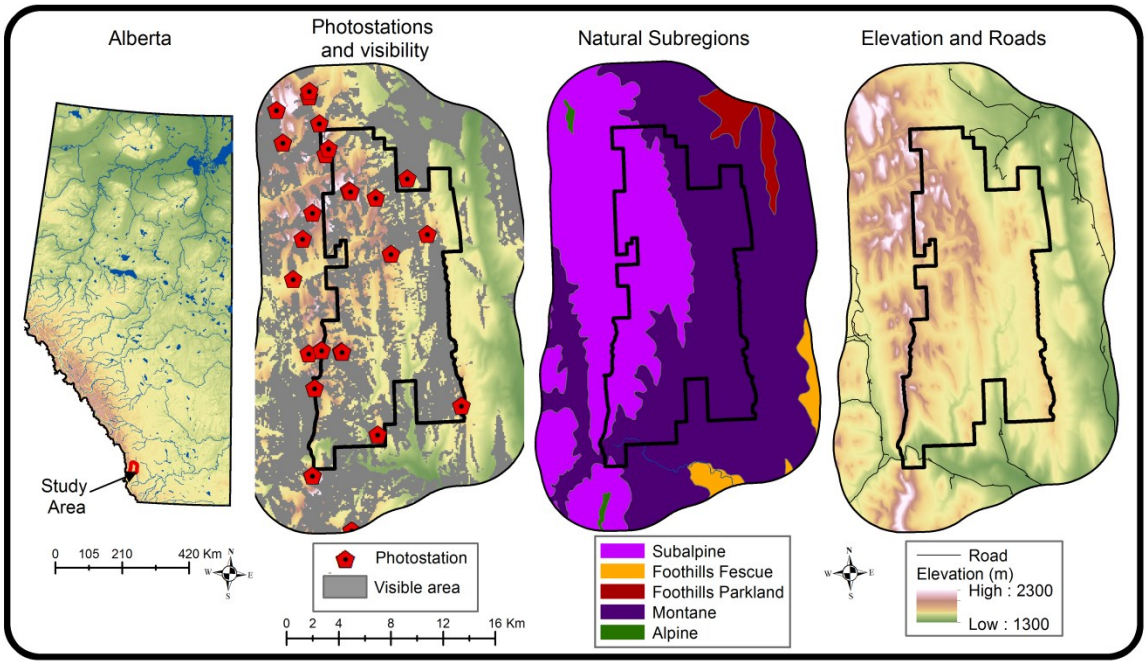

Figure 5.1: Overview of the study area. Bob Creek Wildland (outlined in black) is located in the SW corner of Alberta, and the study area includes a 5km buffer around the protected area boundary. Photostations are shown on the map, as is the area visible from these photostations. Natural Subregions are shown in the third panel, and the fourth panel shows the elevation and roads.

Vegetation in the Subalpine Natural Subregion of the Bob Creek Wildland consists primarily of forest dominated by lodgepole pine, white spruce, aspen, and balsam poplar with lesser components of Engelmann spruce and subalpine fir. The most common shrubs in this subregion are shrubby cinquefoil and creeping juniper. In the Montane Natural Subregion, there are extensive fescue grasslands interspersed by forests dominated by Douglas-fir, aspen, balsam poplar, with limber and whitebark pine on the exposed rocky ridges. The common shrubs in this subregion are bog birch and several species of willow. According to the management plan for the area (Government of Alberta 2011), it is home to approximately 150 bird species, 57 mammal species, two reptile species and four amphibian species. It is one of the most significant overwintering ranges for elk in Alberta, and is the largest intact piece of montane wilderness in the province. The Bob Creek Wildland and adjacent Black Creek Heritage Rangeland are protected under the Alberta Wilderness Areas, Ecological Reserves, Natural Areas and Heritage Rangelands Act, and managed to

“preserve their unique natural heritage, culture and biodiversity in perpetuity for future generations” (Government of Alberta 2011, page 7).

5.3.2. ***Fire Modelling***

To examine the effects of changing the fuel structure in the BCW from its current condition to its historical (1909) condition, we used the Burn-P3 model of the Canadian Forest Service (Parisien et al. 2005) to model burn probability, fire intensity, and fire size across a large landscape (the Calgary Forest Area with a 20km buffer, Figure 5.2). To do so, we needed to understand the locations and probabilities of fire ignitions. The information and data required to run the model on our two scenarios was as follows. The modern (2014) fuel composition of the landscape for the baseline scenario was supplied by the Government of Alberta Forest Protection Branch’s annually updated provincial fuel grid (GOA fuel grid). The historical restoration scenario was based on historic vegetation structure as derived from analysis of photographs taken as part of MP Bridgland’s 1913-1914 Survey (Mountain Legacy Project historical photographs). A detailed digital elevation model obtained from the Government of Alberta supplied topographic information. Modern day fire environment variables included: weather, 1961-2014 ignition locations, fire sizes and duration of burning, and the weather conditions associated with each fire (all supplied by the Government of Alberta Forest Protection Branch). By limiting the modern day fire environment inputs to the 1961-2014 period we limited the potential of the model to recreate large landscape scale fires (such as the 1910 fires), however, we do not know the weather conditions associated with the 1910 fire to recreate such an event. Vegetation in the Burn-P3 model is represented by Canadian Forest Fire Behaviour Prediction System (CFFBP) (Stocks et al. 1989) fuel types. Burn-P3 accounts for changes in plant phenology by using different fire behaviour algorithms depending on whether broadleaf deciduous vegetation has leaves or not, and by “curing” grasses at the appropriate time of year in the model. Furthermore, seasonal changes in fire behaviour are modelled by stratifying the fire environment inputs by season. Fire perimeters that are outputs of the Burn-P3 model do not have any degree of mortality associated with them, but in some cases this can be inferred as each cell that burns has a fire intensity (kW/m) value associated with it. More details on all of this information, data, and model runs can be found below.

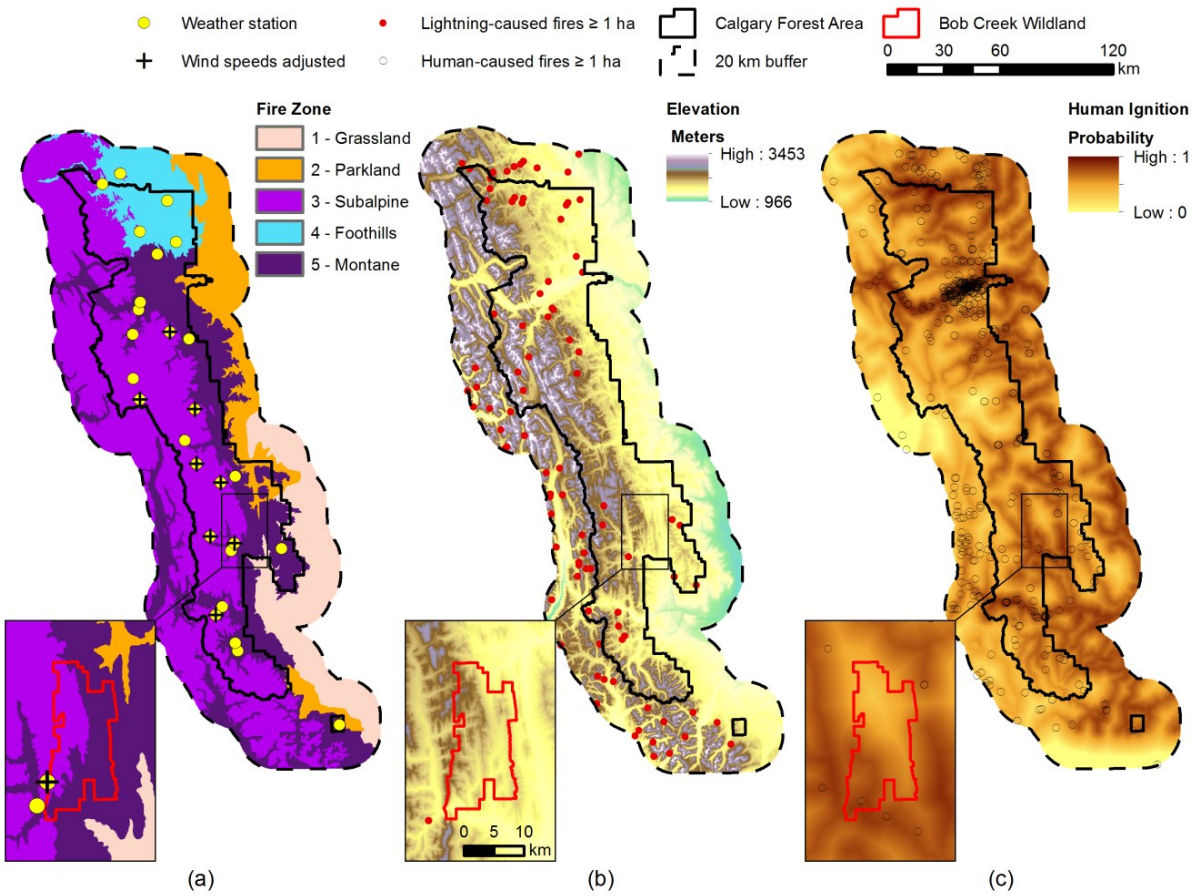

Figure 5.2: Modelling area with Burn-P3 inputs. A) Fire Zones/Natural Subregions with the locations of weather stations. See "daily fire weather" in Table 5.4 for description of weather stations with wind speeds adjusted. B) Elevation and lightning ignition locations from 1961-2014. C) Human ignition probability surface derived from ignition locations 1961-2014. The area outlined in all maps is the Calgary Forest Area with a 20km buffer surrounding it.

5.3.2.1. Baseline Scenario Landscape

The baseline scenario landscape input for the Burn-P3 model was the current (to 2014) fuel grid maintained by the Government of Alberta (GOA). We clipped out the portion of the fuel grid that falls within the Calgary Forest Area and includes the BCW. We also had to obtain fuel grids for the areas that fell outside the Calgary Forest Area that were included in the 20km buffer, and these were obtained from the British Columbia Ministry of Forests and Parks Canada and merged with the Alberta provincial fuel grid. The Alberta grid is derived from the Alberta Vegetation Inventory (AVI) (Resource Information Branch 2005) and is updated on an annual basis to account for disturbances such as forest harvesting, wildfires, and other land use dispositions. The AVI is manually interpreted from air photos,

and has a minimum polygon resolution of 2 ha. The GOA converts the AVI to Canadian Forest Fire Behaviour Prediction System (CFFBP) (Stocks et al. 1989) fuel types using a translation matrix and transforms it into a 1-ha resolution raster grid. The CFFBP fuel types present in the Bob Creek Wildland in the 2014 fuel grid were C1 (spruce lichen woodland), C2 (boreal spruce), C3 (mature lodgepole pine), C4 (immature lodgepole pine), C5 (red and white pine), C7 (Ponderosa pine or Douglas-fir), D1/D2 (leafless/leafy aspen), M1/M2 (leafless/leafy mixedwood), and O1 (grass). These fuel types had to be harmonized with our vegetation classification system, and this procedure is described below.

5.3.2.2. Historic Restoration Scenario Landscape

The historic restoration scenario was limited to restoring the vegetation only within the Bob Creek Wildland to a reasonable representation of its pre-settlement condition. The current fuel grid was used for the modelling area outside the BCW, while the fuel composition within the BCW was replaced based on vegetation classes as visible in the MP Bridgland 1913-1914 Crowsnest Survey photographs; interpolation was used for the areas not visible in the photographs (further details below).

5.3.2.1.1 Image Analysis

To determine the vegetation of the historic landscape we used photographs from the Mountain Legacy Project (MLP) Collection (Higgs et al. 2009) taken by MP Bridgland during his 1913-1914 survey of the Crowsnest Forest Reserve and Waterton Lakes National Park. See Fig. 5.3 for two example images showing the historical and modern day landscape in paired images. These images were some of the ones used in our companion study (Chapter 4) to evaluate vegetation change between 1913-2009 over a larger 320,000 ha landscape which also included the BCW. We used 36 of the photographs from our companion study (Chapter 4) plus an additional seven photographs to provide better coverage of some areas of the BCW (see Fig 5.1 for the locations of the photostations used in this study). A further 18 images were used to fill in a 5km buffer around the Bob Creek Wildland (hereafter BCW5K) for a total of 61 images. Three of these 61 images were from the Sheppard Survey of 1914, as the southeastern corner of Bob Creek Wildland was not surveyed by Bridgland. The historical vegetation in the 5km buffer zone was not used in the historical restoration scenario directly, but was used in the interpolation routines. See Table 5.1 for a summary of the number of images and area visible in the BCW and surrounding BCW5K.

Table 5.1: Number of Mountain Legacy Project photographs used to reconstruct the vegetation of 1909 in the Bob Creek Wildland and surrounding 5km buffer zone.

	Bob Creek Wildland Only (BCW)	Bob Creek Wildland + 5km buffer (BCW5K)
Total Area	20,775 ha	66,053 ha
Visible Area	12,051 ha (58%)	33,338 ha (50.4%)
# Mountain Legacy Images Used	43	61
# Mountain Legacy Images Used from Chapter 4	36	54

We used the WSL Monoplotting Tool (Bozzini et al. 2012) to georeference the images, and followed the procedures outlined in Stockdale et al. (2015, and Chapter 3) to extract raster data. We overlaid a spatially referenced grid over the photographs (Figure 5.3) to classify the vegetation at a resolution of 1-ha/cell in one of seven vegetation classes: conifer- (CF), broadleaf deciduous- (BD) or mixedwood- (MX) forest, shrubland (SH), open canopy woodland (WD), grassland (MG), or non-vegetated (NV). Large fires had burned extensively across the study area and much broader landscape in 1910 (Pyne and Maclean 2008) and were evident in the 1913 images. In addition to classifying vegetation, we wanted to capture recent disturbance on the landscape, and grid cells visibly burned in 1910 were classified as “disturbed fire” (DF). This method failed to detect any fires that burned: a) at low severity causing no visible overstory mortality; b) at high severity that burned all dead wood away completely; d) through grasslands leaving no evidence. To avoid the transient effects of the 1910 fire, we rolled back the vegetation from 1913 to the year before the fire (1909). Visible grid cells with no evidence of fire were assigned the same vegetation category in 1909 that they were in 1913. For visible grid cells with code “DF”, the 1909 vegetation category was determined by the density and form of the standing dead timber in the 1913 images (areas with dense coniferous snags were classified as CF, mixed BD and CF snags as MX, BD snags as BD, and low density snags as WD). These same procedures were used in our companion study of landscape change (Chapter 4).

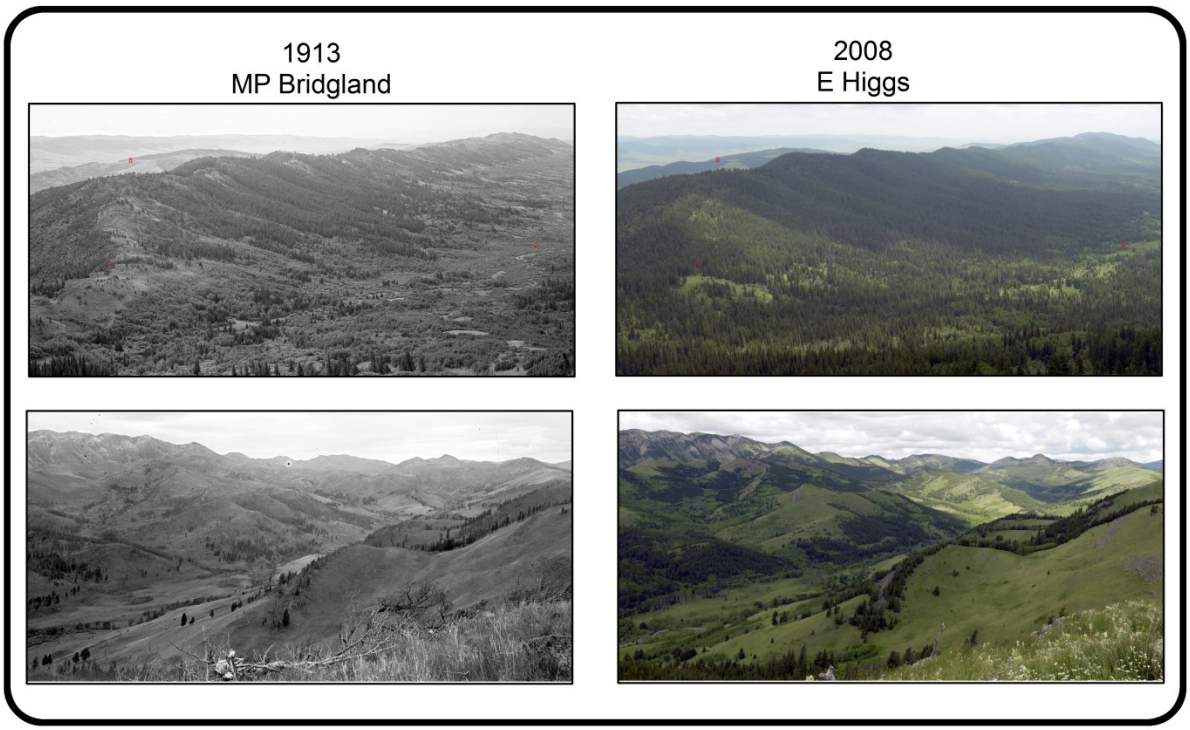

Figure 5.3: Mountain Legacy Project paired photographs from the 1913-1914 MP Bridgland Survey repeated in 2008. The 1913 historical photographs were used to create the historical restoration scenario, and the modern photos are included here only to show an example of the degree of vegetation change between the two time periods.

Despite using 61 images, only 50.4% of the BCW5K was visible in the oblique angle photographs (58% in the BCW itself), which was relatively consistent with what we found in our companion study of the larger landscape (Chapter 4). The area visible in the photographs is referred to as the historic visible landscape, and the grid cells within this visible landscape are referred to as the historic visible grid.

5.3.2.1.2 Interpolation of Non-visible Areas

To model fire behaviour we needed to create a continuous fuel layer from the historic visible grid and thus we needed to be able to interpolate the non-visible portions of the Bob Creek Wildland. Kriging is an interpolation method that first builds a model of the spatial variability of known values, and then uses this model to estimate values at locations whose values are unknown (Li and Heap 2014). We chose indicator kriging (IK) to interpolate the non-visible portions of the landscape as it is one of the only kriging variants for categorical

data (i.e. vegetation categories) as opposed to continuous response variables, and is used in ecology (Wang 2007; Martinez 2013) and geology (Solow 1986; Marinoni 2003) to predict discrete boundaries between vegetation categories and mineral deposits, respectively. The IK procedure tends to perform better than simpler nearest neighbour analyses (Solow 1986; Marinoni 2003; Li and Heap 2014), but requires binary response data rather than multiple categories. For our purposes, the most important distinction was separation of grassland from forest, as the fire intensity within conifer fuel types present in the Bob Creek Wildland varies less than it does between conifer fuel types and grasslands, and the difference between broadleaf deciduous and mixedwoods is dependent upon the whether they have leafed-out or not (Wotton et al. 2009).

In order to validate our use of IK for predicting forest and non-forest portions of the landscape not visible in the historical images we used the provincial fuel grid from 2014 (GOA fuel grid) and conducted two separate tests. First, we used a randomly selected 80% (model building input layer) of the GOA fuel grid to predict what was in the omitted 20%, and compared this prediction to what was actually there (80/20 Test). Second, we masked out the areas of the GOA fuel grid that were non-visible in the historic landscape, and used the visible 58% of the landscape (model building input layer) to predict vegetation in the non-visible 42% of the area and compared the predictions to what was actually there (Visible/Non-visible Test). Although we only needed to fill the gaps in the BCW for Burn-P3 modelling, we used the entire BCW5K zone for interpolation as we felt the interpolation would be more accurate by using a larger landscape to build the predictions. Figure 5.1 shows which parts of the landscape were visible in the images and which parts needed to be interpolated. We did not validate IK beyond testing forest versus non-forest cover.

To conduct our first validation (80/20 Test) of IK for interpolating forest versus non-forest in the GOA fuel grid (2014) we created a binary response variable with a value of 1 for “forest” and 0 for “non-forest” for each 1-ha grid cell. The CFFBP fuel types present in the Bob Creek Wildland in the 2014 fuel grid were C1 (spruce lichen woodland), C2 (boreal spruce), C3 (mature lodgepole pine), C4 (immature lodgepole pine), C5 (red and white pine), C7 (Ponderosa pine or Douglas-fir), D1/D2 (leafless/leafy aspen), M1/M2 (leafless/leafy mixedwood), O1 (grass), and non-fuel. We recoded the GOA fuel grid into forest and non-forest fuel categories (O1 and non-fuel were classified as non-forest, all others were classified as forest).

We used the Indicator Kriging option in the Geostatistical Wizard of ESRI ArcMap 10.4 and used the following parameters and procedures on our model building input layer. A

value of zero was entered as the threshold (the null hypothesis was whether the response variable was forest). We used the semivariogram model with a nugget option to build the model of spatial autocorrelation from which the spatial predictions were made. We ran separate analyses using curve fitting types “stable” and “exponential”, and also ran each model with and without anisotropy (anisotropy implies that the spatial relationship is directionally dependent). Lag size was set to 100m to match the distance between the centroid of grid cells. We used the default search neighbourhood parameters in the Geostatistical Wizard interface of neighbourhood type = standard, maximum neighbours = 5, minimum neighbours = 2, sector type = 4 sectors with 45 degree offset, and the angle, major and minor semi-axes values were copied from the semivariogram calculated above. Thus we ran four separate models (stable-anisotropy; stable-no anisotropy; exponential-anisotropy; exponential-no anisotropy) for the 80/20 test and compared the outputs to determine which model best matched the input layer (see below for description of how the best model was chosen). The best model was then used to predict the 20% of the landscape that was not included in the model building input layer.

Indicator kriging is an inexact form of kriging, meaning the predicted value may differ from the observed value at known locations. The output of the procedure in ArcGIS is a spatial layer with several attributes: a) the observed value of the cell if it was known (for cells to be interpolated this value was blank); b) a probability for each cell of whether it had a value of non-0 (the threshold value); c) a predicted class value (if the cell had a probability value of .5 or greater, then the class value was 1 (forest), and if the probability was less than .5, its class value was 0 (non-forest)). A “mean prediction error” was calculated for all cells in the input layer that were used to build the semivariogram. This mean prediction error is the mean of the differences of all cells’ observed value (binary 0 or 1) and probability (as above, decimal ranging from 0-1). For each model run above (stable/exponential, with/without anisotropy) we recorded the mean prediction error value, and the model with the lowest value was chosen as the best model. The probability value output layer was then used to interpolate what was in the 20% of the surface that was not included in the model building input layer. We compared this predicted 20% of the landscape to what was actually there in the GOA fuel grid to determine the overall accuracy of the method. We repeated the same IK procedure for the Visible/Non-visible test and compared the predicted output for the non-visible layer to what was actually present in the GOA fuel grid to determine the accuracy of the method when there were larger holes to be filled.

Once we had determined the accuracy of IK for predicting the GOA fuel grid, we followed the same procedures described above to interpolate the non-visible portion of the historical (1909) landscape. In total we needed to distinguish 7 different vegetation categories on the historical landscape (forest, CF, BD, MX, WD, MG, and NV); while we analysed shrubs separately in Chapter 4, in this study we merged the SH category with BD as this is how they were classified in the provincial fuel grid. The vegetation categories we used did not match the Canadian Forest Fire Behaviour Prediction System (CFFBP) fuel types (Stocks et al. 1989) required to run the Burn-P3 model. We chose to first interpolate our vegetation categories, and then harmonized these vegetation categories with the CFFBP fuel types. We conducted several IK analyses to separate "contrast pairs": a) forest from non-forest; b) CF from non-CF; c) BD from non-BD; d) MX from non-MX; e) MG from non-MG; f) WD from non-WD. The category of NV was handled in a different fashion and is described below. The WD vegetation class shared characteristics with both forest and meadows and grasslands, and we needed to determine whether it fit best in the forest or non-forest category. We subdivided the contrast pair of forest versus non-forest into two separate contrast pairs to compare the accuracy of including woodlands in the forest category or the non-forest category: a1) forest+WD versus non-forest; a2) forest versus non-forest+WD.

For each contrast pair described above (a1, a2, b, c, d, e, f) we first used 80% of the visible grid cells to build a series of models as we did with the 80/20 and Visible/Non-visible tests above (stable-anisotropy; stable-no anisotropy; exponential-anisotropy; exponential-no anisotropy). We chose the best model based on its mean prediction error as above, and tested its accuracy on the held-back 20% of the visible grid cells. We then used all of the visible grid cells to predict what was in the non-visible grid cells. We ran the IK procedure with the semivariogram curve-fitting and anisotropy options identified in the model selection procedure.

Once we determined that open canopy woodlands were best included in the forest category, we combined all the contrast pairs' IK output layers into a single polygon layer. Each grid cell had an observed vegetation class (if it was in the non-visible grid this value would be missing), and attribute columns with the probability value for being forest, CF, BD, MX, MG, and WD. In all non-visible portions we first selected all cells with a probability of >= 0.5 of being forest, and then assigned the forest type based on the highest probability value from the CF, BD, MX, or WD attributes. Anything that was in the non-forest category (probability of being forest < 0.5) was either MG or NV. We then used the non-fuel cells in

the GOA grid to "stamp out" NV from the non-forest layer (leaving only MG), as these all occur at high elevation in talus slopes or rocky outcrops that we assumed would have changed little between 1909-2014. We also overrode any predictions with a "Water" designation from the GOA fuel (there were only 16 ha of water in the entire BCW area).

Although we interpolated fuels in the non-visible portions of the entire BCW5K area, to create the historic fuel grid for use in the Burn-P3 model we were only interested in "restoring" the fuels in the BCW proper. For the historical fuel grid to use in the historical restoration scenario we used the 58% of the area that had observed values plus interpolated values for the remaining 42% within the BCW only. This historic fuel grid containing both the historic visible landscape plus the interpolated non-visible portions of the historic landscapes is referred to as the "BCW 1909 grid". This BCW 1909 grid was embedded within the GOA grid for the surrounding landscape outside the BCW, and this represented the historical restoration scenario ("1909 restoration grid").

5.3.2.3. Vegetation Classification and Fuel Type Harmonization

In both scenarios, outside of the BCW we used the fuel types that were supplied by the GOA fuel grid, but within the BCW we had to harmonize our vegetation classification system with the GOA fuel grid CFFBP fuel types (Table 5.2) for both scenarios. We compressed some CFFBP fuel types (e.g. coniferous fuel types and mixedwood percentages) as we lacked the detail in the historical photographs to separate some of the vegetation categories into more complex fuel types. Five conifer fuel types (C1, C2, C3, C4, and C5) were all compressed into C3 as it was the most prominent in the study area and these were matched to our CF vegetation class. We matched our MX class to the M1/M2 fuel type, however we did not differentiate mixed conifer – broadleaf deciduous ratios, and therefore set all M1/M2 fuel type cells at 50% conifer / 50% broadleaf deciduous. We matched our BD vegetation class to the D1/D2 fuel type, and MG to O1. We retained C7 as a separate fuel type from the other conifer fuel types as it was generally spatially distinct from the other conifer fuel types, and largely represented Douglas-fir and limber pine growing on rocky ridges and outcroppings through the study area. Our WD class represents a wide range of possible fuel types and therefore was the most difficult to harmonize with the CFFBP fuel types. Areas in the historic visible landscape that were categorized as WD would correspond to fuel type O1 where the trees were very widely spaced, or C7 if the trees were closer together. Given that there is more than 100 years between the two

landscapes, we felt that it was safe to assume anything that was “too open” to be considered C7 in 2014, would have been even more open in 1909, so we called any WD (from 1909) that intersected O1 in 2014 as O1 in 1909 as well. All remaining areas classified as WD were assigned as C7.

Table 5.2: Harmonization of vegetation categories from the historical photography analysis in the left hand column, the Government of Alberta grid of Canadian Forest Fire Behaviour Prediction System fuel types (Stocks et al. 1989) in the right hand column, and the final fuel type used in the Burn-P3 modelling runs in the middle column. C7 = Ponderosa pine/Douglas-fir, C1 = spruce lichen woodland, C2 = boreal spruce, C3 = mature lodgepole pine, C4 = immature lodgepole pine, C5 = red or white pine, D1/2 = broadleaf deciduous (1 is leafless, 2 is leaf-on), M1/2 = mixedwood leafless (1) and leaf-on (2) (% indicates proportion of broadleaf deciduous in the mix, remainder is conifer), O1 = grass.

Historical Photography Vegetation Category	Final Fuel Type for Burn-P3 Modelling	Government of Alberta Fuel Grid 2014
Open canopy woodland (WD) not in 2014 O1	C7	C7
Conifer (CF)	C3	C1,2,3,4,5 M1/2<10%
Broadleaf deciduous (BD)	D1/D2	D1/D2 M1/2>90%
Mixedwood (MX)	M1-50%	M1/2 20-80%
Grassland (MG) Open Canopy Woodland (WD) in 2014 O1	O1	O1
Nonvegetated (NV)	Non-fuel	Non-fuel

5.3.2.4. Modelling Burn Probabilities and Fire Sizes

We used Burn-P3 (Parisien et al. 2005) to model the relative likelihood of fire (hereafter termed "burn probability") and fire size distribution in both scenarios across the modelling area (Calgary Forest Area plus 20km buffer), however we only evaluated the results of the model runs within the Bob Creek Wildland plus the 5km buffer. We chose to include the 5km buffer in our study area for analysis because we expected that changes in burn probability would occur in the surrounding area due to fires burning out of the BCW. We felt that a 5km buffer would capture most of the escaped fires because: a) the western edge of the BCW is composed of non-vegetated high elevation mountain ridges that would prevent fires from burning out of the BCW to the west; b) while fires could burn out of the BCW to the east, they would enter the complex terrain of the Porcupine Hills; and c) with

predominant westerly air flow, long distance fire movement north and south would be limited.

The first management scenario was the baseline scenario, which reflected the landscape of 2014 (the unaltered GOA fuel grid, or “2014 baseline grid”). The second management scenario was the historical restoration scenario with the 1909 restoration grid in the BCW (but not restored in the 5km buffer). Burn-P3 is a landscape-level Monte Carlo simulation model that combines deterministic fire growth modelling of individual fires with probabilistic fire ignition, fire spread, and weather (Parisien et al. 2005). We assembled Burn-P3 inputs using methods described in detail by Parisien et al. (2013). We buffered the Calgary Forest Area out by 20km (hereafter referred to as the “modelling area”, Figure 5.2) to allow fires to ignite outside the area and burn within its boundary. We included the 5km buffer around the Bob Creek Wildland to evaluate the effect of changing fuels within the park on fires that might ignite within the park but burn beyond the park boundary. We only simulated fires greater than or equal to 4 ha, and accordingly, we only used fires meeting this size threshold to develop model inputs. This fire size threshold accounts for 98.5 % of total area burned in the Calgary Forest Area during the period 1961 to 2014 (Government of Alberta 2016). Static and stochastic inputs used to model burn probability are described in Table 5.3, and Figures 5.2, 5.4 and 5.5. Stochastic model inputs were derived from GOA fire records maintained from 1961-present.

Table 5.3: Static and stochastic inputs used to model burn probability using Burn-P3. See also figures 5.2, 5.4, and 5.5.

Model Input	Data Type	Description
Static inputs:		
Fuels	Categorical raster	Canadian Forest Fire Behavior Prediction System fuel type classifications and non-fuel features (Table 5.2) from the provincial 2014 fuel grid. Fuel type classifications were obtained from British Columbia and Parks Canada in the buffer portion of the modelling area that fell outside Alberta's jurisdiction. See Figure 5.8a-b.
Elevation	Continuous raster	Shuttle Radar Topography Mission (SRTM) elevation data re-sampled to 100 m resolution (meters above mean sea level). See Figure 5.2.
Fire zone	Categorical raster	Alberta natural subregions grouped according to mean-annual ignition densities. Biogeoclimatic classifications were obtained from British Columbia and Parks Canada in the buffer portion of the modelling area that fell outside Alberta's jurisdiction. See Figure 5.2.
Wind grids	Continuous raster	The influence of topography on local wind direction (degrees) and wind speed (km/h) as simulated by WindNinja 2.5.4 (Forthofer 2007) for the eight cardinal directions. See Figure 5.2.
Seasons	Setting	Start and stop dates for fire weather, grass curing, and broadleaf deciduous green-up change: - Spring = Apr-1 to May-31 (75 % grass curing, leafless broadleaf deciduous) - Summer = Jun-1 to Aug-31 (40 % grass curing, broadleaf deciduous green-up) - Fall = Sep-1 to Oct-31 (60 % grass curing, leafless broadleaf deciduous) See Figure 5.4a
Stochastic inputs:		
Number of fires	Frequency distribution	Number of fires ≥ 4 ha per year (or iteration). Historical records of the number of fires ≥ 4 ha per year were fitted to a negative binomial distribution. See Figure 5.2, 5.4 and 5.5.
Escaped fire rates	Frequency distribution	Proportion (%) of fires ≥ 4 ha occurring in each combination of season, cause (human, lightning), and fire zone. See Figure 5.5.
Spread days	Frequency distribution	Number of days a fire is expected to spread. Distribution was derived from data from 2000-2015 fire status records for fires < 200 ha, and from Moderate-Resolution Imaging Spectroradiometer (MODIS) hotspot detections for fires ≥ 200 ha using the weighted by mean and distance method described in Parks (2014). See Figure 5.5.
Spread hours	Frequency distribution	The number of hours per day a fire is expected to spread. This input was not derived from empirical data. Burning hours were calibrated so that the distribution of simulated fire sizes was similar to historic fire records for years 1961 to 2014. See Figures 5.6.
Ignition locations	Continuous raster	Relative probability surface of human ignition locations is based on 1961-2014 fire history records and the model assigned ignitions based on these probabilities. Lightning ignitions were located randomly with

		equal probability in all areas stratified by each fire zone by the model. See Figure 5.2.
Daily fire weather	Numeric list	Daily weather conditions observed at noon MST and associated Canadian Fire Weather Index System codes and indices partitioned by season and fire zone. Weather observations from 27 stations with ≥ 10 years of historical records were used. Wind speeds observed at high elevation stations were scaled to mean elevation using a factor of 0.36. The Fine Fuel Moisture Code (FFMC), Initial Spread Index (ISI) and Fire Weather Index (FWI) were recalculated for stations with scaled wind speeds. We then extracted days with fire-conducive conditions using a FWI threshold of 19 or greater (Podur and Wotton 2011). See Figure 5.2.

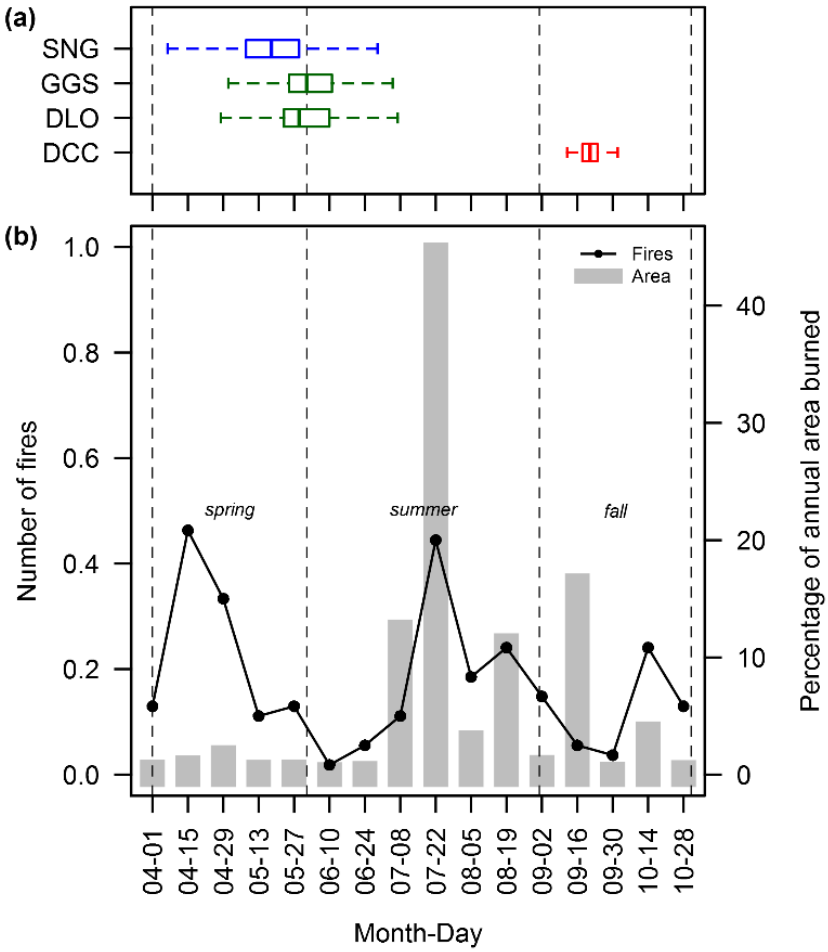

Figure 5.4: Analysis of historical fire records from 1961-2014 in the Calgary Forest Management Area. This shows the mean precentage of annual area burned and annual number of fires for each over 14 day periods from April 1 to October 28. The box and whisker plots in box a) show the mean, standard error, and range in dates associated with SNG = date of snow melt, GGS = grass greenup date, DLO = broadleaf deciduous leafout date, and DCC = fall broadleaf deciduous colour change. These fire records were used to determine the fire frequency distributions (Figure 5.5), and the dates were used to determine fire seasons (see Table 5.3 and Figure 5.5).

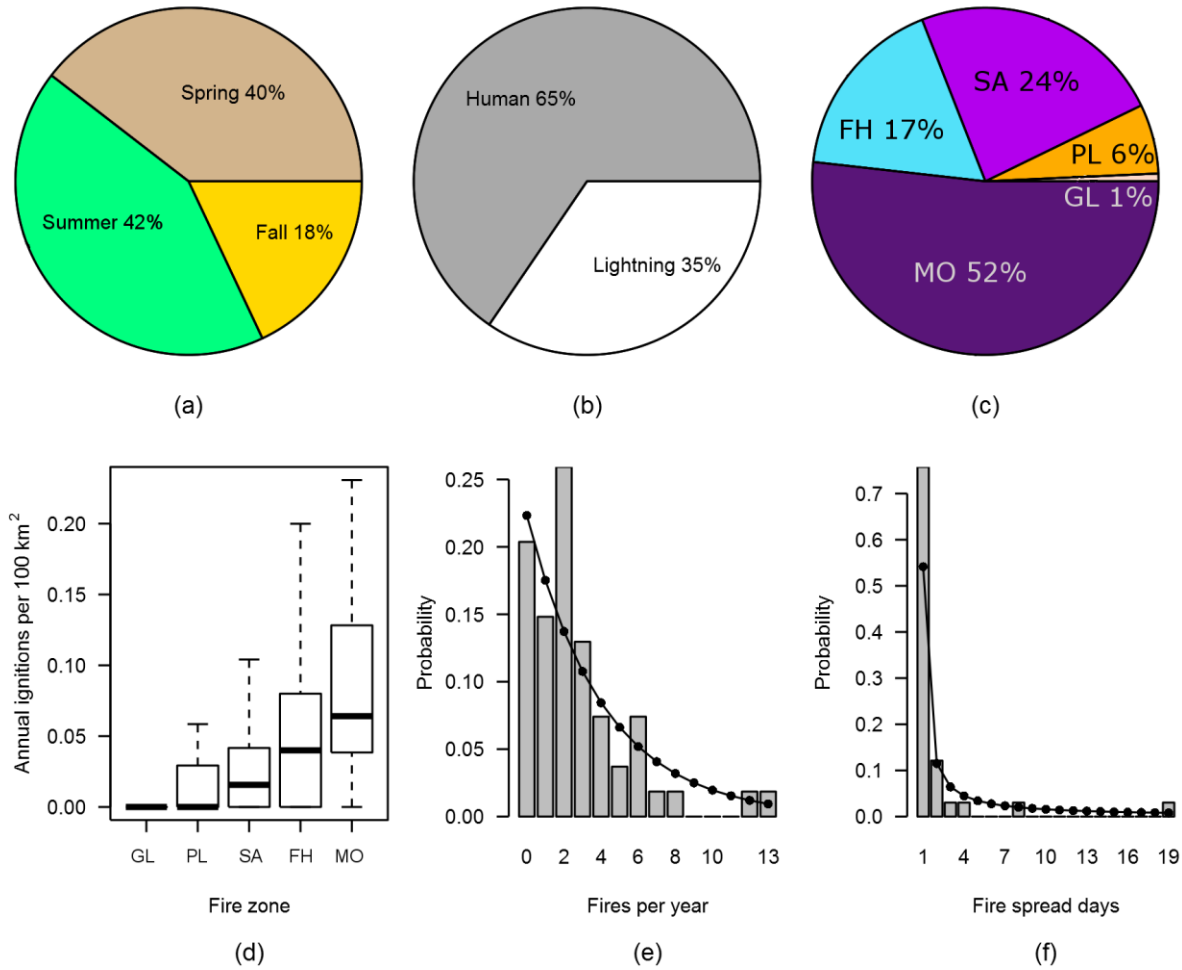

Figure 5.5: Fire history statistics 1961-2014 for the Calgary Forest Area. The inset pie charts indicate a) the percentage of ignitions occurring by season, b) the proportion of lightning versus human caused fires on the landscape, c) the proportion of fires occurring by fire zone (FH = foothills, SA = subalpine, MO = montane, PL = parkland, GL = grassland) Panel d) box plots show median, 25th and 75% percentiles and range of the annual ignitions per 100km^2 in each fire zone. Panel e) shows the probability of fires per year across the landscape, this curve is derived from the same data that created Figure 5.4. The probability distribution was fitted to a negative binomial distribution and is one of the key inputs to Burn-P3 to determine how many fires to burn in each simulation. Panel f) shows the distribution of the mean number of days a fire grows for once it has ignited. This curve is fitted using a gamma distribution and is explained in Table 5.3. This is one of the key inputs to Burn-P3 to determine how long each fire burns in each simulation.

We wanted to control for and hold constant the influence of all variables within the model with the exception of the changed fuel composition within the Bob Creek Wildland. For this reason we used a new feature in the Burn-P3 model called the “replay function”. Using this new feature, all stochastic inputs (locations, date, time of day, weather conditions, duration of burning) associated with each ignition are saved as an ignition table. We ran half of the iterations using the baseline scenario first to create the ignition table, and half of the iterations using the historic scenario to create the ignition table. We then ran a “replay” of these ignitions on the historical restoration scenario and baseline scenarios, respectively, using the associated ignition tables. The only variable that changed between the two model scenarios was the fuel grid itself, and therefore any differences in outputs are solely attributable to the changed fuels.

Previous Burn-P3 simulations conducted by the Government of Alberta in the same landscape showed that burn probability began to stabilize around 70,000 iterations (Neal McLoughlin, *personal communication*). Based on this we chose to run 110,000 iterations or hypothetical fire years for the two scenarios to ensure we would have local stability in the Bob Creek Wildland. We verified that we also achieved model stability by 110,000 runs by plotting the Independent Relative Difference value against the number of iterations (Figure 5.6b). The Independent Relative Difference was calculated as follows. After each 500 iterations, the cumulative burn probability surface was subtracted from the previous burn probability surface (number of current iterations – 500 new iterations), and the mean change in values for all cells was calculated and plotted against the number of iterations, which approaches an asymptote as the model stabilizes.

To determine whether the Burn-P3 inputs (weather, ignition probabilities, numbers of ignitions, days of burning and other parameters) produced realistic output, we examined whether the distribution of fire sizes on the landscape produced by the Burn-P3 model was similar to the distribution of fire sizes in observed history. To do this, we plotted the fire size distribution of all fires burned in the baseline scenario model runs against the 1961-2014 actual fire size distribution (Figure 5.6).

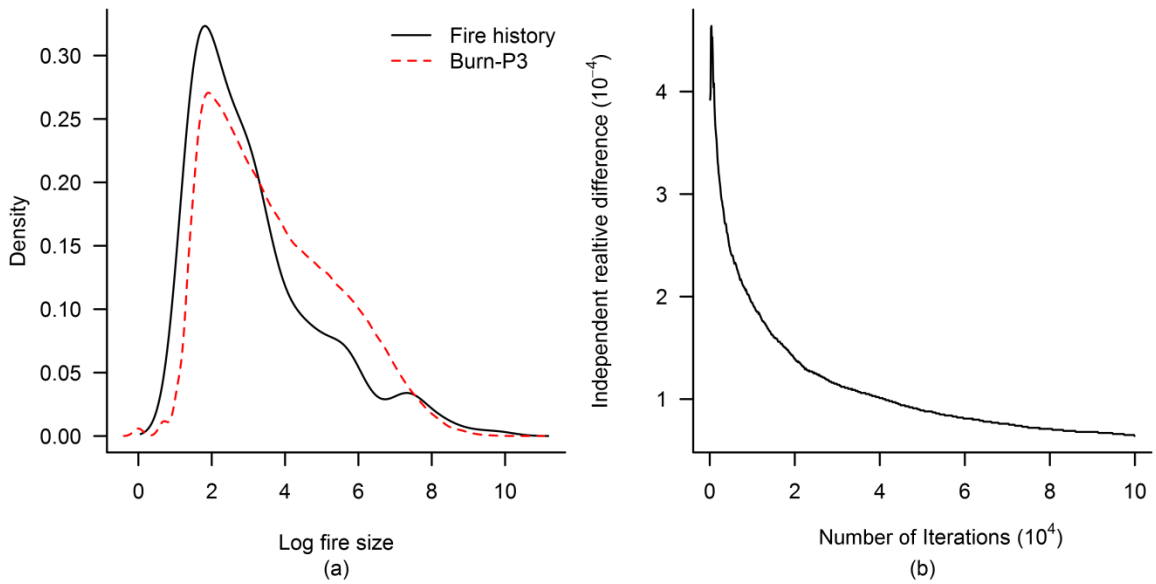

Figure 5.6: Burn-P3 calibration metrics. a) Validation curve showing the distributions of fire sizes in the baseline scenario from the Burn-P3 modelling (red dotted line) and the fire history of 1961-2014. Log fire size is the natural logarithm. b) Measure of the stability of the response surface. Every 500 iterations, the Independent Relative Difference takes the difference of the burn probability of each cell for the cumulative number of iterations minus the previous number of iterations (cumulative – 500 iterations) and calculates the mean. This approaches an asymptote as the model stabilizes.

5.3.3. ***Analysis Methods***

Each iteration of the Burn P-3 model produced a file containing the point of ignition and the perimeter for each fire that burned. Each "burned" grid cell within the burn perimeters has an associated fire intensity (kW/m). We only evaluated grid cells within the BCW5K, however this included any fires that may have started outside this area and burned into the area. These individual layers were added together to count the number of times each grid cell within the BCW and BCW5K burned in the baseline and historical restoration scenario grids separately. The burn probability of each cell (Raw Burn Probability - PB_r) is the number of times each cell burned (burn count) divided by 110,000 (the number of iterations). We also calculated a Conditional Burn Probability (PB_c) for each cell that ws the number of times it burned at an intensity of greater than 4000 kW/m (the level at which direct ground attack is no longer possible). Maps of the PB_r and PB_c were created for both the baseline and historical restoration scenarios.

We calculated the change in the raw burn probabilities (ΔPB_r) by dividing the historical burn count by the baseline burn count. For the ΔPB_c, the calculation was (historical burn count + 1) / (baseline burn count + 1) to prevent divide by zero errors for cells that did not burn at the threshold intensity (4,000 kW/m) in the baseline scenario. We created and visually compared frequency histograms of the raw and conditional burn counts for each scenario in the Bob Creek Wildland (BCW), and the Bob Creek Wildland plus 5km buffer zone (BCW5K).

We created maps showing the ignition point and relative size of each fire that ignited within the BCW5K for each scenario (this excludes fires that ignited outside this zone and burned into it). We created a Δfire_size metric as follows: (historical fire size – baseline fire size) / (historical fire size + baseline fire size).

We compared the fuel grids in both the historic restoration and baseline scenario to measure how much of the fuel grid remained the same, and how much it had changed by fuel type. We calculated how changing the fuel types between the baseline scenario and restoration would affect the Rate of Spread (ROS) differences in fire behaviour using the REDApp Universal Fire Behaviour Calculator (McLoughlin, 2016). For each fuel type in the historical restoration scenario and the baseline scenario we calculated the rate of spread in each of our fire seasons: Spring starts May 1, grass cure = 75%, broadleaf deciduous leaf-off; Summer starts June 16, grass cure = 40%, broadleaf deciduous leaf-on; Fall starts October 1, grass cure = 60%, broadleaf deciduous leaf-off. We used the centroid of the Bob Creek Wildland for the geographic location (49.98^o N, -114.28^o E), and the mean elevation of 1656 m.a.s.l.. We used the Fire Weather Indices representing “high” fire danger in Alberta, which for this location were a Fine Fuel Moisture Code (FFMC) of 90, Build Up Index (BUI) of 75, and a Wind Speed (WS) of 20 km/h. We then calculated a weighted mean ROS for each fuel type by weighting each season’s value by its proportion of average area burned (Spring = 0.03, Summer = 0.75, Fall = 0.22). We then mapped the difference in ROS between the historic restoration scenario and the baseline scenario by showing where the ROS increased, stayed the same, or decreased relative to the baseline scenario ROS value.

5.4. Results

5.4.1. *Interpolation of Non-visible Portion of Landscape*

Using the 80/20 Test in the BCW5K to validate the interpolation predicting forest/non-forest cover in the baseline scenario fuel grid we found that the overall accuracy was 86.6%, and that 91.4% of the forested area and 78.1% of the grasslands in the 2014 GOA fuel grid were correctly predicted. A detailed visual scan of the errors revealed that the majority of these classification errors occurred along forest grassland boundaries, and in areas with patchy cover of both types. The Visible/Non-visible test was used to validate forest/non-forest classification in the non-visible portion of the BCW landscape using the visible cells to build the IK prediction model. Overall, the Visible/Non-visible test predicted forest/non-forest cover with 76.8% accuracy, with correct prediction for 83.9% of the forested area and 65.5% of the grasslands in non-visible areas of the 2014 GOA fuel grid. Relative to the 80/20 Test, it was less accurate as the non-visible gaps were considerably larger than the 20% omitted area in the 80/20 Test. A detailed visual scan of the prediction errors showed the majority of the errors occurred in the larger non-visible portions of the landscape, and that several large non-forested (grassland) areas were predicted to be forest, likely because the visible sample surrounding these large areas was entirely forested. Errors also occurred along narrow grassy valley bottoms that were predicted as forest because they were surrounded by forest. As such, the interpolated 1909 fuel grid likely oversimplifies the larger non-visible areas by making them a uniform fuel type, when in reality the coverage is likely more heterogeneous. Table 5.4 shows these results, and Figure 5.7 shows the areas for which vegetation type was not correctly predicted, overlaid on a satellite photograph of the area.

Table 5.4: Results of the 80/20 and Visible/Non-visible tests to assess the accuracy of indicator kriging (IK) to predict forest/non-forest in the study area using the Government of Alberta 2014 Fuel Grid. The 80/20 test was conducted using a randomly sampled 80% of the landscape to predict what was in the 20% held back, which was then compared to the actual category. The Visible/Non-visible test was conducted using the historic visible grid (58% of the landscape) to predict what was in the non-visible 42% of the landscape, which was then compared to the actual category. The numbers in the cells indicate the number of correctly predicted cells first followed by the total number of cells that actually belong in that category; the percent of correctly predicted cells is given in brackets.

Test	Accuracy Forest Prediction (predicted / actual) (% accuracy)	Accuracy Non-forest Prediction (predicted / actual) (% accuracy)	Accuracy Total (predicted / actual) (% accuracy)
80/20	8,204 / 8,885 (92.3%)	3,266 / 4,348 (75.1%)	11,470 / 13,233 (86.7%)
Visible/Non-visible	6,332 / 6,982 (90.7%)	839 / 1,787 (47.0%)	7,171 / 8,769 (81.7%)

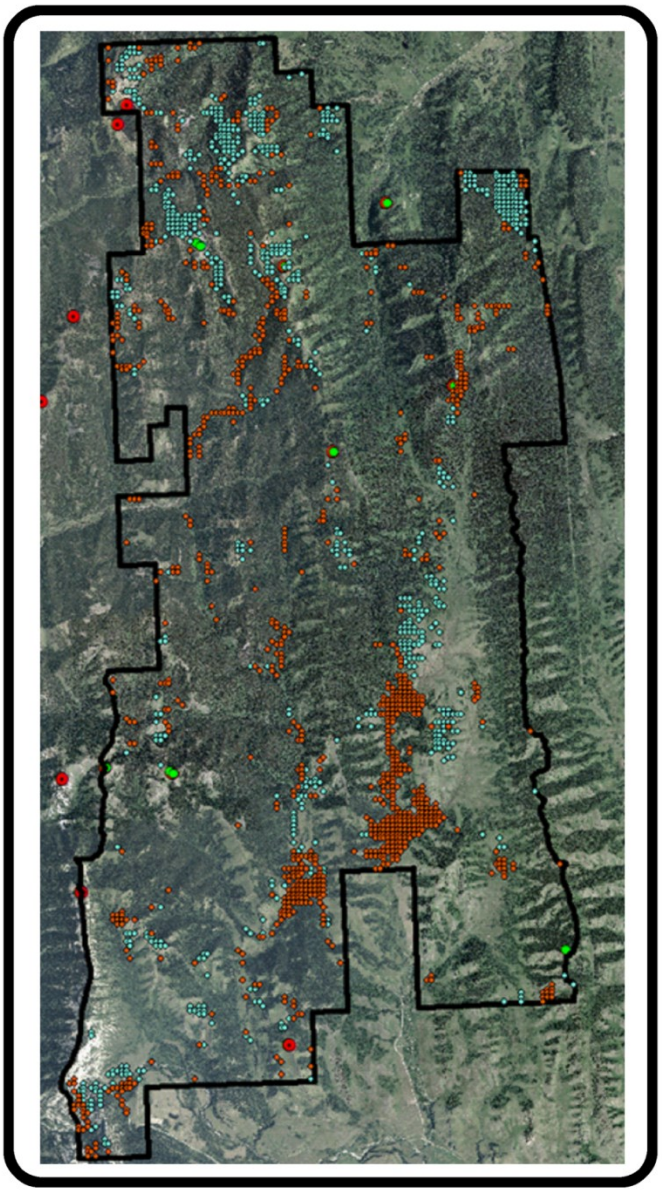

Figure 5.7: Example figure showing the locations where the Indicator Kriging model did not classify forest/non-forest correclty in the Visible/Non-visible test (see Table 5.4) using the 2014 Government of Alberta fuel grid to determine Indicator Kriging accuracy. Red dots indicate areas that were really non-forest but were classified as forest, blue dots are areas that were really forest but were classified as non-forest. The underlying photograph is an orthophoto of the region from 2008.

The locations of open canopy woodland were more accurately predicted when it was included in the forest category than by including it with grassland (Table 5.5). We found the overall accuracy of IK for predicting contrast pairs in the historical restoration scenario grid was good for most categories, however it became much less accurate with predicting MX

and WD forest cover, likely due to their low total cover of the landscape. The results of the indicator kriging for each contrast pair are shown in Table 5.5, and the details of each model used are described further in Appendix C. The fuel input layers to the two scenarios are shown in Figure 5.8a-b.

Table 5.5: Results of the Indicator Kriging interpolations used to fill the non-visible portion of the landscape for use in the historical restoration scenario. These analyses represent the best interpolation models chosen for each contrast pair, and were initially constructed using 80% of the historic visible grid in the Bob Creek Wildland plus the 5km buffer area to predict the vegetation category in the remaining 20% of the historic visible grid. These models were then used to predict the non-visible portion of the landscape in the Bob Creek Wildland only. Each test is composed of a "class member" and "non-class member" (i.e. CF vs non-CF test, CF = class member, non-CF = non-class member). Key model parameters shown include the type of curve that was fit to the variogram model (exponential or stable), and whether the spatial pattern in the contrast pair showed anisotropy or not. The numbers in the cells indicate the number of correctly predicted cells first followed by the total number of cells that actually belong in that category.

Test	Key Model Parameters	Accuracy Class-member Prediction (predicted/actual) (% accuracy)	Accuracy Non-Class-member prediction (predicted/actual) (% accuracy)	Accuracy Total (predicted / actual) (% accuracy)
(Woodland + Forest) vs Grassland	Exponential Anisotropic	3,390 / 3,865 (87.7%)	2,158 / 2,803 (77.0%)	5,548 / 6,668 (83.2%)
(Woodland + Grassland) vs Forest	Exponential	1,880 / 2,678 (70.2%)	3,420 / 3,990 (85.7%)	5,300 / 6,668 (79.4%)
CF vs non-CF	Exponential Anisotropic	2,145 / 2,555 (83.9%)	3,730 / 4,113 (90.6%)	5,875 / 6,668 (88.1%)
MG vs non-MG	Exponential	2,155 / 2,803 (76.8%)	3,390 / 3,865 (87.7%)	5,545 / 6,668 (83.1%)
BD vs non-BD	Stable Anisotropic	675 / 1,038 (65.0%)	5,375 / 5,630 (95.4%)	6,050 / 6,668 (90.7%)
MX vs non-MX	Stable Anisotropic	87 / 272 (32.0%)	6,320 / 6,396 (98.8%)	6,407 / 6,668 (96.0%)
WD vs non-WD	Stable	175 / 633 (27.6%)	5,939 / 6,035 (98.4%)	6,114 / 6,668 (91.7%)

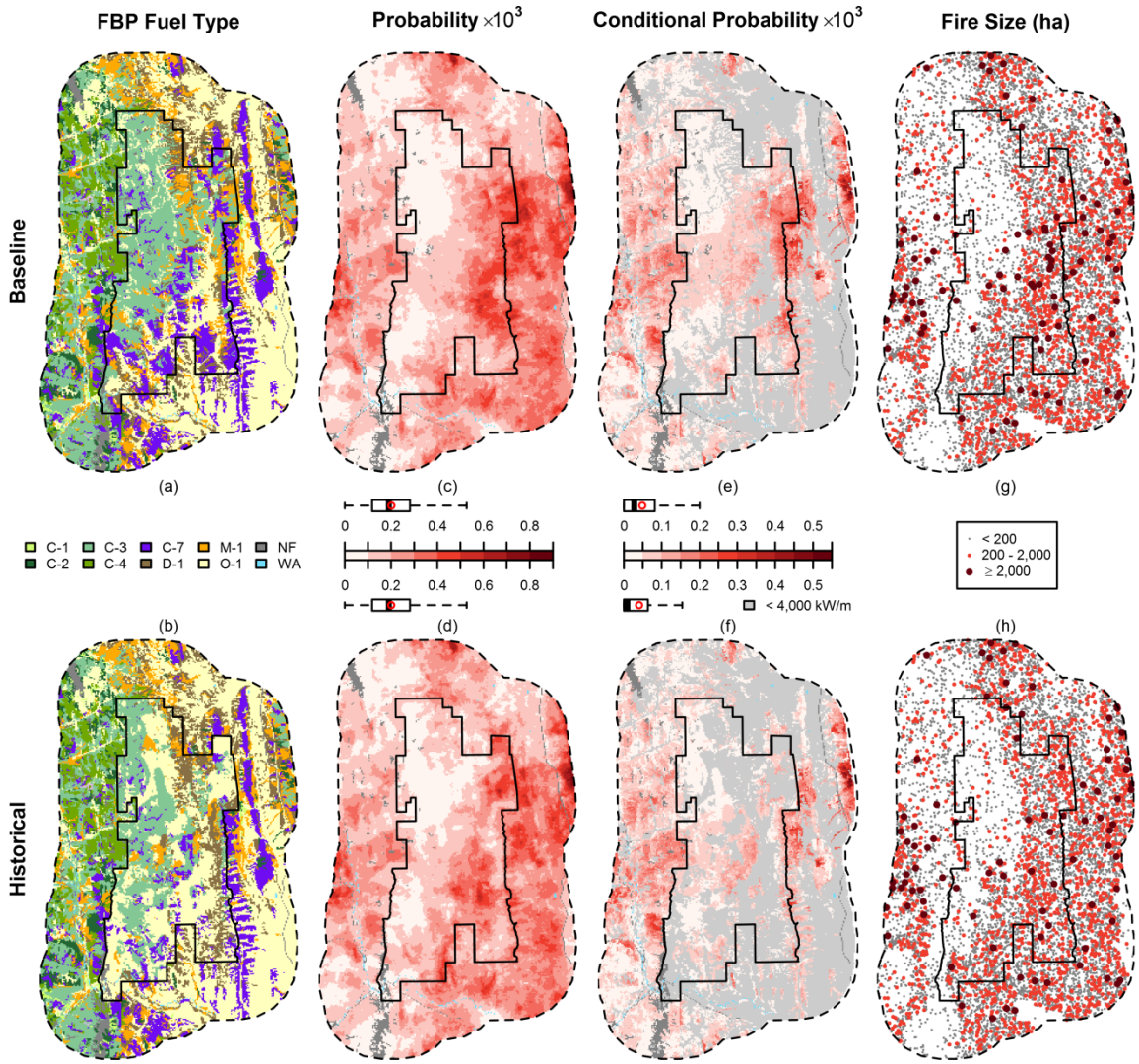

Figure 5.8: Burn-P3 outputs with input fuel grids. Top row panels are from the baseline scenario, bottom row panels are from the historical restoration scenario. Panels a-b show the input fuel grids with Canadian Forest Fire Behaviour Prediction system fuel types in the Bob Creek Wildland plus 5km buffer zone (BCW5K). Panels c-d show the raw burn probability (fire at any intensity). Panels e-f show the conditional burn probability (fire burning >= 4,000 kW/m intensity). Panels g-h show each modeled ignition that occurred within the BCW5K and its associated size class. The box and whisker plots for the probability maps align with the probability values in the legends. The red circle in the box indicates the mean, the middle line in the box represents the median, the edges of the box are the 25th and 75th percentile, and the dotted lines extend to 1.5 x interquartile range.

5.4.2. ***Fuel Changes 1909-2014***

For the visible portion of the landscape in the BCW, 48% had the same fuel type in the historical restoration scenario as it did in the baseline scenario. Since 1909, the BCW landscape saw increases in cover of C3 (from 29.9% to 36.4%), C7 (from 7.9% to 18.1%), and M1 (from 5.7% to 13.3%) fuels, with declines in O1 (from 40.3% to 20.4%) and D1 (from 16.0% to 10.9%) fuels. The most stable fuel type between the two periods was C3, with 70.6% of the C3 fuel in 1909 remaining as such in 2014. The remaining 29.4% of the C3 fuel from 1909 had become O1, M1, or D1 by 2014. Other fuel types converted to C3 over this time, however, so that the total cover of C3 fuel increased by 21.7% relative to the amount of the landscape occupied in 1909. The least stable fuel type over the 1909-2014 period was M1, with only 12.3% of the M1 from 1909 still in the same fuel type in 2014. Of the 16% of the landscape covered by broadleaf deciduous forests in 1909, only 27.5% of it has remained the same fuel type in 2014, and 56% of it had converted to coniferous or mixedwood forest by 2014. Of the O1 fuel type from 1909, 34% remained as O1 in 2014, with much of it converting to C3, C7, M1 and D1 in declining order. Other fuel types converted to O1 fuel between 1909 and 2014 to reduce the overall losses of O1 cover to only 50%. These values and the transition pathways are shown in Table 5.6a.

Table 5.6a: Changes in fuel types in the Bob Creek Wildland 1909 historical restoration and 2014 baseline scenarios. “Totals” column and row indicate the proportion of the landscape covered by each fuel type in each time period, changes are indicated in each transition cell. Numbers in each cell indicate the proportion of the total landscape going through each transition and add to 1. Non-italicized numbers in the top of each cell are for the visible portion of the landscape (observed), and the italicized numbers at the bottom of each cell are for the total landscape (visible plus interpolated). D1-D2 = leafless/leafy aspen , O1 = grassy, C7 = Douglas-fir, C3 = Lodgepole pine (and all other conifers), M1-M2 = leafless/leafy mixedwood (Stocks et al. 1989).

Historical Restoration Scenario Fuels	Baseline Scenario Fuels						
	Nonfuel	D1-D2	O1	C7	C3	M1-M2	Totals Historical
Nonfuel	**0.002** ***0.008***	0 *0*	0.001 *<0.001*	<0.001 *<0.001*	0.001 *0.001*	<0.001 *<0.001*	0.004 *0.009*
D1-D2	<0.001 *<0.001*	**0.044** ***0.030***	0.027 *0.020*	0.016 *0.013*	0.029 *0.025*	0.044 *0.033*	0.160 *0.121*
O1	0.004 *0.003*	0.041 *0.042*	**0.137** ***0.137***	0.079 *0.076*	0.094 *0.124*	0.049 *0.048*	0.403 *0.430*
C7	0 *0*	0 *0*	0 *0*	**0.079** ***0.075***	0 *0*	0 *0*	0.079 *0.075*
C3	0.003 *0.002*	0.015 *0.014*	0.035 *0.040*	0 *0*	**0.211** ***0.228***	0.032 *0.029*	0.299 *0.313*
M1-M2	0 *<0.001*	0.001 *0.007*	0.003 *0.004*	0.007 *0.006*	0.029 *0.027*	**0.007** ***0.006***	0.057 *0.050*
Totals Baseline	0.009	0.109	0.204	0.181	0.364	0.133	**1**

When we look at the full BCW landscape (including the interpolated portions), we saw relatively consistent total cover values for each fuel type in the visible and non-visible portions of the landscape, and in the transitions rates for each fuel type between the two scenarios. We saw slightly higher levels of O1 and C3 fuels in the full historical restoration scenario grid than in the visible portions only, and slightly lower levels of D1, M1, and C7.

5.4.3. ***Burn-P3 Results***

With the exception of areas that were non-vegetated, all portions of the landscape within the BCW5K burned at least once in these simulations and the mean number of times any given cell burned was 22.6 in the historical restoration scenario (s.d. = 11.9, maximum =

94) and 22.8 in the baseline scenario (s.d. 12.5, maximum = 92). A total of 10,881 fires ignited within the BCW5K in the 110,000 Burn-P3 iterations (3,059 in the BCW, and 7,221 fires within the 5km buffer zone). The largest fires originating in the BCW grew to 8,259 ha and 6,538 ha in the baseline and historical restoration scenarios respectively. For fires originating in the 5km buffer zone these sizes were 32,219 ha and 20,276 ha in the baseline and historical restoration scenarios respectively (these fire sizes include the full perimeter of the fire, including area outside the BCW5K). There were 531 "did not burn" ignitions, which were fires that failed to reach the 4 ha size threshold in the baseline scenario but did reach that threshold in the historical scenario. Conversely, there were 515 "did not burn" fires in the historical scenario (failed to reach 4 ha in the historical scenario, but reached that threshold in the baseline scenario). The maximum fire size achieved in the historical scenario that did not burn in the baseline scenario was 387 ha, and conversely the maximum fire size achieved in the baseline scenario that did not burn in the historical scenario was 762 ha.

The mean area predicted to burn at any intensity (the raw burn probability, PB_r) within the BCW in any given iteration (the sum of all individual cells' burn probabilities) under the range of conditions assumed in this model was 3.71 ha in the baseline scenario and 3.66 ha in the historical restoration scenario. These values changed to 13.23 ha and 13.18 ha respectively when including the 5km buffer zone (see Figure 5.8 c-d). Considering only the conditional burn probability (PB_c, mean fire intensity >= 4,000 kW/m) the overall burn probability of the landscape was lower, and much of the baseline and historical restoration scenario landscapes did not burn at all at this intensity (compare Figure 5.8c to 5.8e, and Figure 5.8d to 5.8f to see the change from PB_r to PB_c). The mean area predicted to burn at intensities exceeding 4,000 kW/m in any given year under the range of conditions assumed in this model was 1.05 ha in the baseline scenario, 0.58 ha in the historical restoration scenario, and these were 3.2 ha and 2.65 ha respectively when including the 5km buffer zone (Figure 5.8e-f). The change in PB_c between the two scenarios represented a much larger reduction (44.8% within the BCW, 17.2% reduction within the BCW5K) than what was seen when comparing the reduction in PB_r in the historical restoration scenario relative to the baseline scenario (1.3% reduction in the BCW, and 0.3% reduction in the BCW5K).

The areas of the landscape with the highest PB_r in the baseline scenario and the historical restoration scenario were along the eastern edge of the BCW along the Whaleback Ridge and the valley immediately to the west of this ridge. Burn count values (PB_r * 110,000 iterations) ranged from 20-30 (baseline) and 10-30 (historical) in the valley

bottom to between 30-60 (baseline) and 10-50 (historical) along the ridge. Relatively higher PB_r was also observed within the buffer zone to the east where burn count values were all above 20. Some areas within the eastern buffer had burn count values of 40-50+ (baseline) and 20-50 (historical), and the Porcupine Hills had spots with burn counts higher than 70 in both scenarios. The areas with the lowest PB_r were along the western edge of the BCW, which is mainly along the eastern slopes of the Livingstone Range. These areas tended to have burn count values below 10 in both scenarios.

While the overall mean PB_r did not differ much between the two scenarios in the BCW, and even less in the BCW5K, there were visible differences between the two scenarios (see Figure 5.9b). Along the eastern edge of the BCW the PB_r was considerably lower in the historical restoration where it was only 50-90% of what it was in the baseline scenario. In some smaller areas the PB_r was reduced to less than 20% of what it was in the baseline probability (Figure 5.9b). Most of these historical restoration scenario reductions in PB_r occurred on parts of the landscape where the fuels had been changed from conifer (M1, C3, C7) in the baseline scenario to broadleaf deciduous (D1) in the historical restoration scenario (see Figures 5.8a-b, and look at the locations of the brown colouring (D1) fuels in Figure 5.8b). This was observed primarily in the north-central zone of the BCW, and along the western aspects of the Whaleback Ridge along the eastern border of the BCW. The fires that originated in these areas became smaller in the historical restoration compared to the baseline scenario (Figures 5.8g-h, 5.9d as shown by the concentration of blue dots along the eastern edge of BCW). These areas with decreased PB_r in the historical restoration scenario were offset by other large areas of the BCW with 1.1 - 2 fold increases in PB_r relative to the baseline scenario. These areas of increased PB_r coincided with areas that were changed to O1 fuel from other fuel types (see areas in Figure 5.8a that are turquoise (C3) or purple (C7) that became large areas of tan (O1) in Figure 5.8b). Some smaller scattered areas showed three to five-fold increases in PB_r relative to the baseline scenario. The total area within the BCW that had increases in PB_r in the historical restoration scenario relative to the baseline scenario was 49.8% of the landscape (42.6% in BCW5K), with 9.6% the same between both scenarios (20.6% in BCW5K), and 40.6% with a lower PB_r in the historical restoration scenario (36.8% in BCW5K).

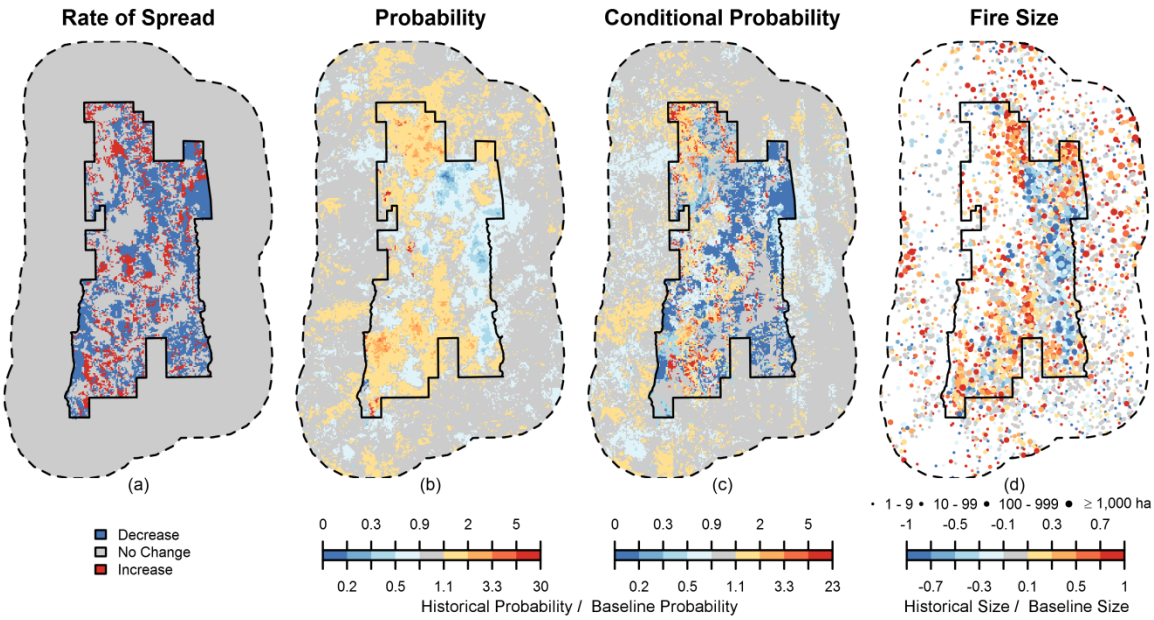

Figure 5.9: These maps show the differences (delta values) between the two modelling scenarios using the 2014 scenario as the baseline. Blue indicates decreases and red indicates increases from the baseline scenario due to the changed fuel types in the historical restoration scenario. Panel a) shows the net change in rate of spread associated with particular fuel type changes. Panels b) and c) show historical burn count divided by baseline burn count for fire at any intensity and at intensity >= 4,000 kW/m respectively. In panel c, "1" has been added to all burn counts to control for divide by zero errors. Note: b and c are plotted on different scales, and gray background indicates no change. Panel d) shows changes in fire sizes for each ignition ((historical – baseline) / (historical + baseline). The points in d) are scaled in size relative to the the absolute value of historical – baseline fire sizes.

The differences were much greater between the two scenarios with regard to the conditional burn probability (cells that burned at an intensity greater than 4,000 kW/m) (Figure 5.9c). The area with the greatest change in PB_c was in the northeast corner of the BCW where the PB_c was less than 20% what it was in the baseline scenario. These reductions, like with reductions in PB_r were concentrated in the north central valley, along the Whaleback Ridge, and in the southeast corner where fuels had been changed from C3, C7 or M1 in the baseline scenario to D1 in the historical restoration scenario. The reductions occurred over a greater portion of the landscape than they did with the changes in raw burn probability, as the areas with reduced PB_c also included the areas that had been converted to O1 fuel in the historical scenario (Figure 5.9c). There were some areas (northwest and southwest corners) where the PB_c was higher in the historical scenario. Overall, the area of

the BCW for which the PB_c increased in the historical restoration scenario relative to the baseline scenario was in 22% of the landscape (17.9% in BCW5K), 30.4% was the same between the two scenarios (52.4% in BCW5K) while 47.6% of the landscape had a lower PB_c in the historical restoration scenario than in the baseline (29.6% in BCW5K) (Figure 5.9c).

In areas that converted from C3 or C7 (baseline) to O1 (historical) (see areas in Figure 5.8a that are turquoise (C3) or purple (C7) that became large areas of tan (O1) in Figure 5.8b) fire sizes became larger (see the coincidence of red and orange dots in Figure 5.9d with these expanded areas of O1 fuels in Figure 5.8b). In general the fire sizes for the historical restoration scenario were smaller than in the baseline scenario with mean, median and maximum values of 146.6 ha, 13 ha, and 8,739 ha respectively in the historic restoration compared to 192.6 ha, 23 ha, and 10,070 ha in the baseline scenario.

The effect of changing the fuels inside the BCW in the historical restoration scenario also affected the burn probability outside the BCW (recall that the fuels were not changed outside the BCW). We saw that in the 5km buffer zone around the park there was a considerable amount of area with a higher PB_r (see Figure 5.9b) in the historical restoration scenario than in the baseline, with only a small amount of the area in the buffer zone showing lower PB_r in the historical restoration scenario. We saw similar results in the change in PB_c (see Figure 5.9c). This influence likely extended well beyond this 5km buffer, but we did not evaluate this any further from the BCW border. We also saw that fires originating in the 5km buffer zone outside the BCW tended to be larger in the historical restoration as compared to baseline scenario (Figure 5.9d).

We examined how the fuel changes would affect the expected rate of spread (ROS) of fires occurring in a given grid cell on the landscape. We saw that of the 51.6% of the landscape that had changed fuels between 1909 and 2014, 37% of the landscape would burn at lower ROS while 14.6% would burn with a higher ROS in the historical restoration scenario as compared to the baseline scenario (Figure 5.9a). For fires burning in the spring, the rate of spread in O1 fuels increased sharply. All changes in rates of spread expected with each fuel transition are shown in Table 5.6b.

Table 5.6b: Changes in expected rates of fire spread (Δmeters/minute) in fuel type transitions in the Bob Creek Wildland 1909 between the historical restoration and 2014 baseline scenarios. Bold numbers at the top of each cell indicates the weighted mean change in rate of spread for all fires regardless of season, the bottom row of numbers in each cell indicate changes in rate of spread associated with Spring/ Summer/ Fall fires. D1-D2 = leafless/leafy aspen , O1 = grassy, C7 = Douglas-fir, C3 = Lodgepole pine (and all other conifers), M1-M2 = leafless/leafy mixedwood (Stocks et al. 1989). Bottom row of the table is weighted mean rate of spread (m/min) in each fuel type (top bold value), with values for spring / summer/ fall below.

Historical Restoration Scenario Fuels	Baseline Scenario Fuels				
	D1-D2 (Δm/min)	O1 (Δm/min)	C7 (Δm/min)	C3 (Δm/min)	M1-M2 (Δm/min)
D1-D2	**0**	**-2.21** -14.69/ -1.22/ -3.87	**-2.64** -0.64/ -3.31/ -0.64	**-6.44** -4.44/ -7.11/ -4.44	**-8.27** -7.26/ -8.61/ -7.26
O1	**2.21** 14.69/ 1.22/ 3.87	**0**	**-0.44** 14.05/ -2.09/ 3.23	**-4.24** 10.25/ -5.89/ -0.57	**-6.07** 7.43/ -7.39/ -3.39
C7	**2.64** 0.64/ 3.31/ 0.64	**0.44** -14.05/ 2.09/ -3.23	**0**	**-3.80** -3.80/ -3.80/ -3.80	**-5.63** -6.62/ -5.30/ -6.62
C3	**6.44** 4.44/ 7.11/ 4.44	**4.24** -10.25/ 3.80/ 0.57	**3.80** 3.80/ 3.80/ 3.80	**0**	**-1.83** -2.82/ -1.50/ -2.82
M1-M2	**8.27** 7.26/ 8.61/ 7.26	**6.07** -7.43/ 7.39/ 3.39	**5.63** 6.62/ 5.30/ 6.62	**1.83** 2.82/ 1.50/ 2.82	**0**
Rate of Spread	**1.34** 3.34/ 0.67/ 3.34	**3.54** 18.03/ 1.89/ 7.21	**3.98** 3.98/ 3.98/ 3.98	**7.78** 7.78/ 7.78/ 7.78	**9.61** 10.60/ 9.28/ 10.60

Within the BCW, we saw that there was little difference in the distribution in the raw burn counts between the historical and baseline scenarios, although there were more cells that burned with higher frequency in the baseline scenario (Figure 5.10, bottom right panel). When looking at the difference in distribution of conditional burn counts (> 4,000 kW/m intensity) in the two scenarios, we saw that the baseline scenario clearly had more cells burning more frequently than in the historical restoration scenario (Figure 5.10, top right panel). The differences in the raw burn counts were not as pronounced in the BCW5K zone as they were in the BCW (Figure 5.10, bottom left panel), however we saw there were much higher conditional burn counts in the baseline scenario (Figure 5.10, top left panel).

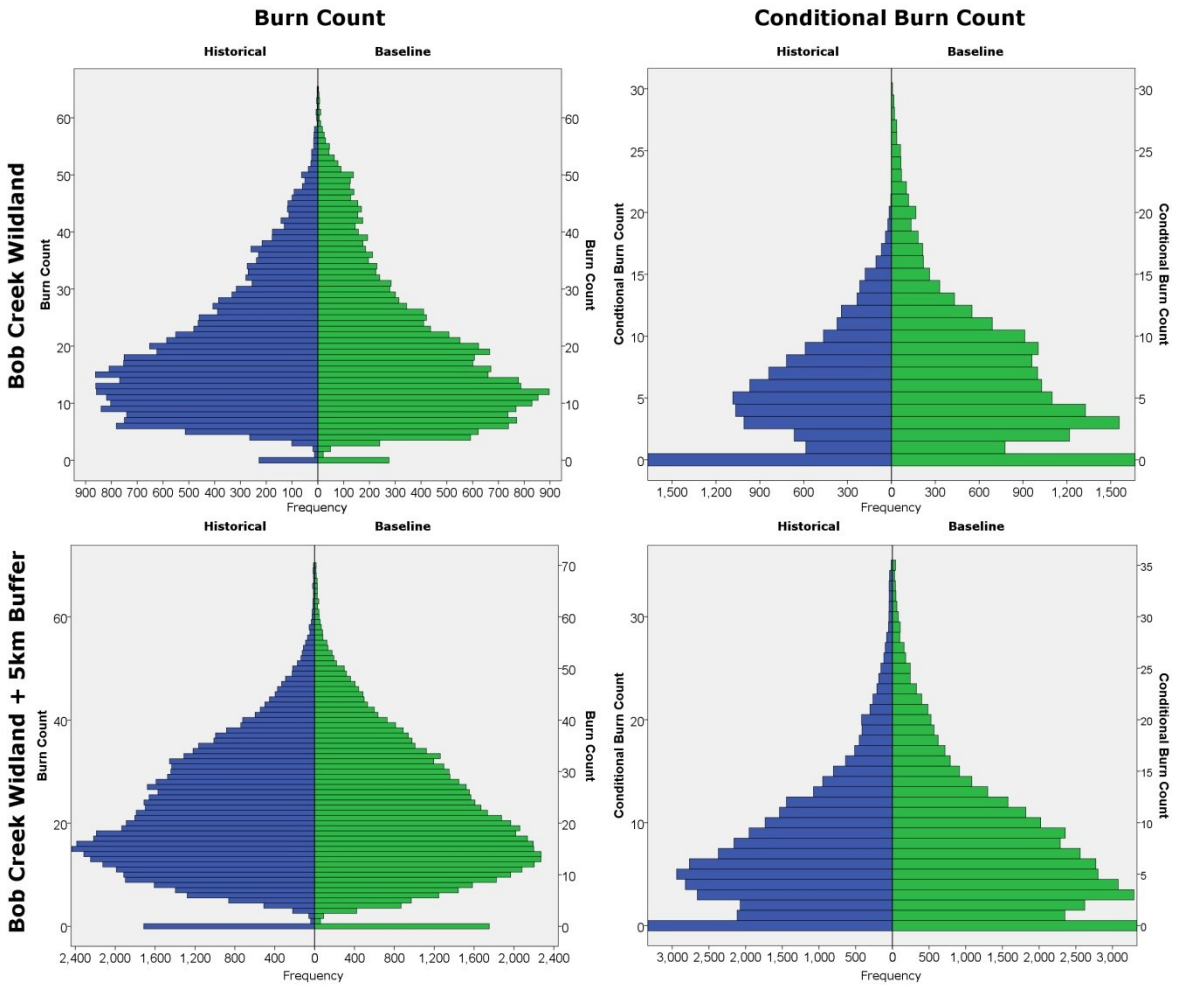

Figure 5.10: Frequency distributions for number of fires that occurred in each cell (burn counts) from the Burn-P3 model scenarios. Each panel shows the frequency distributions for the historical restoration (blue) and baseline (green) scenarios. The top row shows the distributions within the Bob Creek Wildland only, and the bottow row shows the distributions in the Bob Creek Wildland plus the 5km buffer zone. The first column of panels shows the burn counts per cell for all fires, and the second column shows the conditional burn counts per cell for fires burning at greater than 4,000 kW/m intensity. In the right column panels the “0” frequency bars extend to well out of visible range. In the top right panel, the 0 count values are 11,151 and 6,142 for the historical and baseline, respectively. In the bottom right panel these values are 31,999 and 26,866 for historical and baseline, respectively.

5.5. Discussion

5.5.1. *Changes in Burn Probability in the Bob Creek Wildland*

By using historical photographs to determine what the historical vegetation composition the landscape was at the turn of the 20th century, we were able to test the assumption held by many fire management agencies today that restoring this vegetation structure would reduce the probability of high intensity, large landscape-scale wildfires from occurring. As we found, the historical restoration scenario resulted in very little difference in the mean raw burn probability of the landscape (fires of any intensity), and some might think that such a negligible reduction in mean raw burn probability indicates that changing fuels would not be worth the considerable expense and effort for a management agency. However, focusing only on the mean raw burn probability of the landscape ignores the larger issue of wildfire risk, which is defined as a combination of the likelihood of fire (burn probability), the intensity of the fire, and the consequences of a fire occurring (Miller and Ager 2013). By examining these three together we can create a more complete picture of how the wildfire risk to the ecological values of this landscape was reduced with this historical restoration scenario. While the mean raw burn probability changed very little between the two scenarios, the historical restoration scenario nearly halved the mean conditional burn probability (fires with intensity greater than 4,000 kW/m). In many parts of the landscape the conditional burn probability was reduced to less than a tenth of that in the baseline scenario. Mean fire size was also considerably lower in the historical restoration scenario. We also saw changes in the spatial pattern of raw and conditional burn probabilities on the landscape; some areas showed elevated burn probability, but these were offset by other areas that showed decreases in burn probability. To account for these changes in wildfire risk due to reductions in burn probability, fire intensity, and fire size, we need to look at how changing fuel types affected fire behaviour and examine what the consequences of these changes would be.

In the two scenarios we examined we held constant the number, location, and timing of ignitions, duration of burning, and the weather conditions under which the fires burned. Thus, the only way the historical restoration scenario affected burn probability was by changing the speed at which fires moved across the landscape (rate of spread). For the purposes of this discussion, fuel changes that resulted in increased or decreased rates of spread are considered as fire “accelerants” or “suppressants”, respectively. Whether a particular fuel transition acts as an accelerant or suppressant is not just a product of the fuel

type itself, but varies depending on the seasonality of the fires. The largest fuel change was that 30% of the landscape changed from forested fuels in the baseline scenario to grassland in the historical restoration scenario, and the overall effect of this change on burn probability is complex and difficult to predict. We know that fires burning in grass in the spring and fall can burn very large areas (Rowe 1969; Bailey and Anderson 1980; Brown et al. 2005). Cured grassy fuels (O1) have higher rates of spread in the spring compared to any forest fuels present in the study area (Rothermel 1983; Taylor et al. 1997; Wotton et al. 2009), and fires starting in, or burning through the increased amounts of grasslands in the historical restoration scenario would have grown much larger (Finney 2001). In the summer, however, when O1 is actively growing and green, the C3 (mature lodgepole pine), C7 (Douglas-fir) and M2 (mixedwood leaf-on) fuel types all have higher rates of spread, and therefore grass would act as a suppressant compared to these fuels. So too in the fall, O1 would be a suppressant relative to C3 and M1 (mixedwood leaf-off), however it is an accelerant compared to C7. Thus, the net effect of this large shift to O1 fuel on burn probability would depend upon: the spatial location of these changes relative to other fuel types (Miller and Urban 2000); and how much larger the spring and fall fires burned relative to summer fires. The amount of area burned in these seasons between 1961-2014 account for 3%, 22% and 75% of the area burned, respectively. However, due to the fuel changes in the historical restoration scenario the area burned by season may be considerably different, but was not analyzed. The other fuel changes that could be considered accelerants were areas restored to C3 from O1 (3% of landscape), which would have resulted in higher rates of spread in the summer and fall, and areas restored to M1-M2 from C3 (another 3% of the landscape), which increases rate of spread regardless of season. All other accelerant changes were very minor in terms of area affected.

When we look at the maps showing changes in burn probability and changes in expected mean rate of spread for fires (Figures 5.9a-d), some interesting patterns emerge. Areas of the landscape that had fuel changes that acted as fire suppressants would have offset some of the increased burn probability that was the result of higher amounts of grassy fuels and other accelerants in the historical restoration scenario. In addition to the suppressants mentioned above, areas that were converted to D1 and D2 (broadleaf deciduous leaf-off and leaf-on, respectively) from all other fuel types would also have acted as suppressants. The D2 fuel type cuts the rate of spread by 80-90% compared to C3, C7 and M2 fuels, and by 65% compared to O1. D1 (leaf off) fuels are not as strong a suppressant relative to the forested fuels, but are even more effective at reducing the rate of

spread in O1 fuels (by 81%) than D2. As such, these larger areas of broadleaf deciduous in the historical restoration scenario would have significantly reduced the rate of spread of many fires. While these fuel changes to D1-D2 only covered roughly 9% of the landscape, they primarily occurred along the western slope of the Whaleback Ridge, which makes up the eastern border of the study area and is the prevailing downwind side of the Bob Creek Wildland. Fires starting in, or upwind of the areas that were restored to broadleaf deciduous would have been considerably smaller than they would have been had they started in C3, C7, M1-M2, or O1 fuels.

Given what we would expect to see with regard to how fuel changes affect rates of spread, it was not surprising that the areas of the landscape that showed increased probability of burning at any intensity in the historical restoration scenario were largely in or near the areas that were changed to grasslands (O1) or other accelerants. These areas were predominantly in the southwestern, south central, and northwestern areas of the Bob Creek Wildland, with a large pocket in the northeastern corner. The location of suppressant fuel changes appeared to be highly influential. Fires starting in or on the upwind side of areas restored to broadleaf deciduous appear to have been dampened as they reached these slower burning fuels. Increasing the suppressants on the downwind side of most fires seems to have been effective at reducing the overall burn probability of the landscape as we can see that there was a decrease in the burn probability in eastern portion of the 5km buffer zone outside the BCW. The only area where there were significant increases in PB_r in the buffer zone were in the north where there were more accelerants near the edge of the BCW.

The second component of fire risk to be considered is the fire intensity. In terms of changes in the conditional burn probability (i.e. fires burning at an intensity of greater than 4,000 kW/m) we saw very large changes throughout much of the BCW, with less impact within the 5km buffer zone. Whereas the occurrence of new grassland accelerants seemed to increase the overall raw burn probability in many parts of the landscape, the conditional burn probability was reduced in these same areas as well as in the areas restored to broadleaf deciduous. While the increased rate of spread in areas converted to grasslands increases the overall fire sizes, and therefore also raw burn probabilities, the reduction in the conditional burn probability can be attributed to O1 and D1 fuels burning with considerably lower intensity than C3, C7, or M1 fuels (Bailey and Anderson 1980; Taylor et al. 1997). Large reductions in the conditional burn probability (i.e. of intense wildfires) were observed over large areas of the Bob Creek Wildland, but were especially concentrated in the eastern

half of the Wildland, and in a broad band from the middle of the western edge running north east towards the long central valley adjacent to the Whaleback Ridge (see large areas of dark blue in Figure 5.9c).

While fire in grasslands is easier to suppress or manage due to lower intensities, the increased rate of spread in these fuel types in spring and fall can offset this ease of suppression very quickly if there are coniferous forests or values at risk nearby that could ignite from the grass fires. In areas where the response time for suppression efforts may be slower than the potential speed at which grass fires could percolate to coniferous fuels, the overall and conditional burn probability of the landscape could increase. However, suppressing grass fires is not always easier, because under conditions with extreme winds and low humidity grassland fires can be virtually impossible to contain and will rapidly spread to more intense-burning fuels. In the historical restoration scenario, having suppressant fuels on the downwind side of the wildland appeared to be effective in limiting large fires. However, as Miller and Urban (2000) found, under extreme weather conditions and low fuel moisture, the time since the last fire (which affects the amount of fuel available) and the spatial arrangement of fuels are largely irrelevant and the landscape is almost completely burnable. Given that less than 15% of the landscape had mean accelerant fuel changes in the historical restoration scenario (but this varied considerably by season), it appears that they have a disproportionate effect on burn probability as nearly 50% of the landscape had a higher raw burn probability. Similar to what we found for areas converted to grass having a large effect on increasing burn probability (and therefore area burned), Miller and Urban (2000) modelled the relationship between area burned and fuel connectivity, and found that total area burned was strongly and positively correlated to the amount of grass (accelerants) in the fuel bed. To determine the effects of accelerants versus suppressants on burn probability in our scenarios we would have to partition our analysis of the fires in the two scenarios by season, which would require setting up modelling parameters in a different way than we did for this study. It would also be worth examining the effects of accelerants and suppressants in isolation and in combination as additional scenarios.

The third element to examine regarding changes to wildfire risk relates to the consequences of fires. The Whaleback Ridge that makes up the eastern edge of the Bob Creek Wildland (and the western edge of the adjacent Black Creek Heritage Rangeland) is classified as an *Environmentally Sensitive Area* of national importance (Government of Alberta 2011), and there are numerous stands of whitebark (*Pinus albicaulis*) and limber pine (*P. flexilis*), both of which are listed as endangered species in Alberta. The Whaleback

Ridge is also considered one of the two most important winter ranges for elk and other ungulates in Alberta (Government of Alberta 2011). Our modelling scenarios showed that the area within the Bob Creek Wildland and surrounding landscape with the highest burn probability, and which is most likely to burn at the highest intensity was along this ridge and in the valleys to the east (Figure 5.8). This is also the area where the historical restoration scenario resulted in the greatest reductions in the raw and conditional burn probabilities.

Other studies that have examined the effects of changes in landscape fuel arrangement have shown similar results to ours. Wang et al. (2016) investigated the potential effects of a changing climate on burn probability in south-central British Columbia. They included the effects of the changing climate on fuel structure, which caused fuel changes not dissimilar to those we examined (fuels that were less flammable and had lower rates of spread), and, like us, saw overall reductions in burn probability due to changing fuels alone. Fulé et al. (2004) modeled changes in potential fire behaviour between 1880-2040 in the Grand Canyon. They found that in some sites with more than 99% Ponderosa pine, the crown bulk density (an indication of crown canopy closure) had nearly doubled between 1880-2000 (similar to the changes from open canopy to closed canopy forest observed in our historical restoration scenario). In these sites, the crowning index (windspeed required to maintain an active crown fire) decreased from ~140km/h to roughly 65 km/h. This effect of increased crown closure on the crowning index essentially means that the conditional burn probability (i.e. intense fire) would be far lower in the open canopy stands than in the closed canopy stands, which would burn much more easily. Finney (2001) modeled the effects of different landscape level fuel treatment on large wildfires and found that the rate of spread of a head fire could be slowed by using suppressant fuels in the right locations.

5.5.2. ***Model Assumptions and Spatial Interpolation***

As with most modelling exercises, the reliability and accuracy of the outputs are related to the quality of the input data used. The model outputs showed that in the current fire environment, an average 3.71 ha per year is expected to burn in the Bob Creek Wildland, compared to 3.66 ha per year if we were to restore it to its historical condition in 1909. These values are based on the modern fire management activities that have limited the number and size of fires on this landscape over the past 50+ years. These expected rates of burning translate to fire cycles (number of years to burn an area equal to 100% of

the study area) of 5,600 Years and 5,676 years, or mean annual burn rates of 0.0179% and 0.0176% of the landscape. These values are very close to what Tymstra et al. (2005) observed in Alberta for the entire Montane and Subalpine Natural Subregions where the mean annual area burned was less than 0.02% of the landscape between 1961-2002. Given that we used the same data sources to calibrate the Burn-P3 model that Tymstra et al. (2005) used to conduct their study, we can see that Burn-P3 models the area burned quite reliably.

Studies on fire regimes in the landscape neighbouring and including our study area (Hawkes 1979; Arno 1980; Barrett 1996; Rogeau 2005b) show fire cycles prior to the year 1900 were in the range of 15-30 years in the Montane and 30-150 years in the Subalpine. The mean annual burn rate of the landscape that would be expected under these fire cycles would range from 0.67 – 3.3% of the landscape in the Subalpine to 3.33 – 6.67% in the Montane. Clearly, the burn rates we saw in the Burn-P3 model are nowhere near these values. In the larger study area we examined in Chapter 4 (320,000 ha) which included the Bob Creek Wildland, we found that the mean annual area burned since 1913 was 0.075% (a fire cycle of 1,333 years), which is higher than the Burn-P3 model outputs and Tymstra et al.'s (2005) study, but still well below the pre-1900 burn rates. Evidently, even in the latter half of the 20th century the burn rate of the landscape has declined considerably compared to the first half.

This may lead some to question the validity of using historical fire data from 1961-2014 to drive the Burn-P3 model because the fires in this time period have burned in the context of the "suppression era" and do not capture the full range of historic variability in the fire regime. While this does indeed have an impact on the absolute values of burn probability on the landscape, our main questions related to the relative differences between the two scenarios, and these changes were what we focused our discussion on. Furthermore, our intent was not to evaluate how the current landscape and the landscape hypothetically restored to its condition in 1909 would burn under the historic fire regime. Instead, we wanted to evaluate how they would burn in the current fire environment which includes the heavy suppression efforts across the landscape that have reduced burn rates to levels far below the historical mean.

We must also look at the imprecision of the interpolation method used to create the historic landscape. Our objective was not to perfectly recreate what the landscape looked like in 1909, but to create a reasonable representation of what the historic landscape would have looked like in terms of fuels. Given that 58% of the landscape within the BCW was

visible in the historic photographs, the imprecision due to interpolation applied to 42% of the landscape. The indicator kriging procedure predicted the forest/non-forest delineation of the historic landscape with roughly 90% accuracy; thus, the overall landscape delineation of forest/non-forest (grasslands, or O1 fuels) in the historical restoration scenario was roughly 96% accurate. Within the forested areas, the interpolation of C3 fuels (mature lodgepole pine) was the most accurate, with declining accuracy for D1 (broadleaf deciduous), M1 (mixedwoods), and C7 (Douglas-fir).

The largest non-visible areas of the landscape were the areas where the interpolation oversimplified the landscape by placing broad swathes of one fuel type where there likely would have been pockets of different fuels, and these mostly resulted in homogenous grasslands. A visual examination of the modern imagery of the area and the Mountain Legacy Photographs show that most of the grasslands in this area had aspen copses in the areas with higher moisture (depressions, north and east aspects).In the even wetter areas along streams, creeks, and in swales there was also willow growing. The interpolation we used did not produce these features, and if it had, there would have been many more “suppressant” patches within the grasslands. Given that areas converted to grasslands were an accelerant on the landscape in the historical restoration, if the interpolation procedure had placed more broadleaf deciduous vegetation in patches and bands in the grasslands, this would have lowered the PB_r even further in the historical restoration scenario than the model results showed.

These errors due to interpolation likely had some impact on the overall burn probability differences between the two scenarios, and a sensitivity analysis whereby we test different fuel type amounts and spatial arrangements would be an important next step if such a restoration plan was ever enacted. However, if such a restoration effort were to proceed, even if it had a perfectly recreated vegetation map from 1909 to guide efforts, it would take many years and be replete with imprecision due to the unpredictable outcomes of prescribed fire and wildland fire management, and vegetation succession following silvicultural treatments. Given the importance of D1-D2 acting as fire suppressants, it may be not be important to place the D1-D2 fuel exactly where it was located historically. Instead, it might be more effective to place this fuel type in locations to limit high intensity fires occurring around areas of high value and to prevent fires from escaping the Bob Creek Wildland and burning into private lands outside the protected area.

5.5.3. ***Historical Restoration of Landscapes***

Why should we be concerned if the landscape is changing from more open forest cover, broadleaf deciduous forests, and grasslands to closed canopy forests with a higher proportion of coniferous trees? One main reason is that climate change projections suggest that the vegetation trends should be moving in the opposite direction (Schneider 2013; Wang et al. 2016). Because of these two opposing trends, the likelihood of having the vegetation most suited to future climate in the right places will be diminished. We could manage this ecological tension by recognizing that large-scale wildfires will inevitably occur on this landscape and the vegetation best adapted to the site will re-establish following this eventual disturbance. This could occur at any point in time in the future. It is possible that after large tracts of pine-spruce forests are consumed by fire, they will be recolonized by Douglas-fir, limber pine, and whitebark pine if those are indeed the species better adapted to the climate at that time (areas currently classified as Subalpine converting to Montane). We could also see that areas covered by Douglas-fir, limber pine, and whitebark pine today might be replaced by Foothills Fescue vegetation if that is what is best adapted to the site (Montane areas converting to Foothills Fescue). However, for this to occur, a seed source must be available for this conversion, and with the conditional burn probability being much higher for the current landscape vegetation configuration, these seeds may be consumed by fire. Furthermore, Lang and Halpern (2007) showed that when trees were removed from areas where they had previously encroached on montane meadows, the seed bank of meadow species was short lived and the recolonization process was slow.

If managers were interested in conducting a historical restoration by beginning to convert some of the forests to grassland today to lower the probability and severity of future wildfire, they might wish to consider some of the following suggestions. Bai et al. (2004), Sankey et al. (2006), Halpern et al. (2012), and Noss (2013) reviewed and described a variety of approaches and provided guidelines for restoring a grassland system with and without fire. In short, those sites at greatest risk of loss (by examining locations with the highest PB_r and PB_c values) could be triaged and prioritized for either treatment or protection. Prescribed burns could be lit in the grassy valleys, allowing the fire to burn into neighbouring forested stands, removing seedlings, saplings, and “pushing back” the forest edge. It would be wise, however, to first establish suppressant fuels on the downwind side before increasing the landscape abundance of accelerants. Grazing pressure could be manipulated in these new grasslands to either prevent or encourage aspen sprouting, and

semi-regular reapplications of fire could be used to burn out and exhaust the tree seedbed. Some forest thinning could reduce crown bulk density to reduce fire intensity for more active fire control of the forested areas.

5.6. Conclusion

In this historical restoration simulation, we saw only minor reductions in the overall burn probability of the landscape, but large reductions in the likelihood of high intensity fire. By cutting the likelihood of fire burning at high intensity in half, future fires would be easier to manage, severity would be considerably lower, and it would be much easier to protect the biological legacies in the Bob Creek Wildland. However, historical restoration is not a panacea that could be applied everywhere to reduce wildfire risk as the relative position of accelerant versus suppressant fuels is critical. In our case with the Bob Creek Wildland the suppressant fuels were downwind of the accelerants, but the results would have likely been quite different had we not “restored” the downwind side of the wildland to broadleaf deciduous vegetation and instead left this area covered largely by coniferous forest as it is today. If this had been the case, the fires coming out of the grasslands would then enter fuels that would burn at high intensity, propagating fire even further beyond the border of the BCW. Instead, the broadleaf deciduous fuels on the downwind side essentially acted as a fire “brake” and stopped many fires from leaving the Wildland.

Over the past century we have seen large shifts away from grasslands and open canopy forests in montane landscapes in favour of closed canopy coniferous forests throughout wide areas of western North America, and specifically in the Bob Creek Wildland. Given climate change projections for the region, restoring the historical vegetation in all or part of this landscape would increase the chances of having the most suitable vegetation on site for future climate change, and help to maintain the ecological integrity of the largest single tract of Montane wilderness in Alberta. Burn probability modelling is one of the best ways to reveal what portions of the landscape are most at risk to future fires (Miller et al. 2008), and the scenario we tested here confirms that the historical landscape vegetation composition and arrangement would indeed reduce wildfire risk to the ecological values of the Bob Creek Wildland and immediately surrounding area. While some may feel that active intervention to preserve wilderness is undesirable or dangerous, in the absence of intervention, the Bob Creek Wildland is likely to continue to lose grasslands to forest encroachment and this poses a grave risk to the Montane wilderness.

Our focus was not on the complex issue of how land managers would actually accomplish restoration of a 20,000 ha wildland park to its historical condition. Enacting a 20,000+ ha restoration plan that involves changing the vegetation of half the landscape would require significant coordination of silviculture, wildland fire management, prescribed burns, and grazing management. Managers also need to be cognizant of the downwind effects of such restoration actions, because by modifying fuels, land managers could affect the patterns of burn probability across a much broader landscape than the area of concern itself (Parisien and Moritz 2009; Parisien et al. 2010). Using the approach outlined here, managers could evaluate the benefits of historical restoration in any area of ecological concern in general. By using burn probability modelling and testing different vegetation configurations, managers can determine which parts of their landscapes are at greatest risk, and prioritize those parts of the landscape to treat and modify to reduce wildfire risk to ecological values.

Chapter 6: Conclusion and Recommendations

6.1. Conclusion

I am often asked why it matters to investigate ecological history and to determine what processes and events created the landscape we live in. The answer is simple: unless we can understand how our present landscape emerged from our actions in the past, we have no way of predicting what our current actions will create for the future. In this dissertation I created new methods that allowed me to use historical photographs to provide the most detailed description to date of ecological change from the turn of the 20th century to the present day. I was able to extract spatially referenced data from the historical photographs and use this to describe the changes in vegetation cover since the time of European settlement. I then related these changes to anthropogenic disturbance on the landscape, wildfire history, and topography. While I used southern Alberta as my study area, I showed that the changes that have occurred in vegetation cover are consistent with what other studies using very different methodologies have found over much broader geographic areas of western North America. I found that while much of the landscape was in the same vegetation category in 2008 as it was in 1909, that where it has changed it has shifted strongly in favour of coniferous forest at the expense of grasslands and meadows, open canopy woodlands, and deciduous forests. I also tested the assumption of many land management agencies that restoration of historic landscape vegetation structure would lower wildfire risk and found that it would indeed lower the probability of high intensity wildfire, but would have little effect on the probability of fires of any intensity burning. The methods and approach taken in this dissertation could be used as a framework to conduct future studies using historic repeat photography throughout the world. While oblique angle photography is really only useful in complex terrain where images can capture a significant amount of area, there are many mountainous and hilly regions of the world with photographs dating to the late 1800s and early 1900s that could be examined.

In the second chapter of this dissertation I looked at the END model of forest management. Emulation of Natural Disturbance (END) is a variant of ecosystem management that recognizes the importance of disturbance for maintaining ecological integrity. For END to be a successful model for forest management we need to describe disturbance regimes and implement management actions that emulate them. While there has been a considerable amount of research into fire regimes in western North America, they are highly variable through time and this makes choosing reference conditions to

emulate very difficult. There are also numerous knowledge gaps regarding how this variability of fire regimes affects vegetation. Oftentimes, land managers “borrow” information from fire regime studies conducted somewhere else, and while we cannot always have “perfect information”, we need to be cognizant that the fire regime operating in one place may be substantially different than one operating in another. In Chapter 2 I reviewed the components of fire regimes (cause, frequency, timing, extent, and magnitude) and discussed which ones we know the most about, which ones we know the least about, and which ones are the most important to emulate. I outlined some difficulties with using fire regimes as coarse filters for forest management, which included: a) not fully understanding the interactions between fires and other disturbance agents; b) assuming fire is a strictly exogenous disturbance that solely exerts top down control of forest structure; and c) assuming that by replicating natural disturbance patterns we will also preserve ecological processes.

In Chapter 2, I also outlined numerous knowledge gaps regarding what we know about fire regimes. One of these is that we do not know the spatial and temporal extent of mixed severity fire regimes. Mixed severity fire regimes leave much more complex vegetation composition on the landscape, both in terms of species composition and vertical and horizontal structure. Without knowing the spatial extent and temporal bounds of mixed severity fire regimes, we really do not know what the structure left behind after the disturbance looks like. The second knowledge gap is that we know very little about low-severity surface fire regimes in both grassland and forested ecosystems. A third knowledge gap is that we do not fully understand the effects of modern fire suppression and exclusion on fire regimes and landscape vegetation structure. Suppression/exclusion have affected fire regimes in the 20th century, but by how much? It is widely suspected that suppression/exclusion have created conditions that will cause future fires to be larger and burn at higher intensity, but we have little evidence for this. A fourth knowledge gap is that the majority of fire regime studies lack adequate temporal depth to describe their temporal variability. Finally, we also lack detailed spatial coverage for many attributes of fire regimes. In some parts of the landscape we have detailed fire regime information, but in other locations we have essentially no information. I closed Chapter 2 by highlighting how the END model could be strengthened if we could fill some or all of these knowledge gaps. Firstly, we should conduct more research to determine the extent and location of mixed severity fire regimes. Secondly, we could use prescribed burning following harvest to ensure the chemical effects of fire are also incorporated in the END model. Thirdly, we could use

silvicultural techniques to ensure that the differences between post-harvest and post-fire community structure are minimized. Fourthly, we need to increase the temporal depth of our understanding of the interactions between wildfire and ecosystem structure by using more paleoecological techniques, dendrochronological analysis of dead wood, historic climate modelling, and using historical photographs to measure landscape change and investigate past fire effects. Finally, we can continue to invest in extant research programs to experiment on different methods of emulating natural disturbance. The END model shows considerable promise as a management tool, but we need to strengthen the base upon which it is built.

Chapter 3 focused on assessing the WSL Monoplotting Tool to examine historical photographs and creation of new GIS methods to rapidly classify vegetation change. The Mountain Legacy Project is a treasure trove of information regarding the ecological history of the Alberta Rocky Mountains (among other areas), but to the point at which I started this research, these photos had been used only for examining small areas of the landscape, and largely in a non-spatial and non-quantitative fashion. By using the WSL Monoplotting Tool, and creating a new method to extract raster data from oblique angle images, I was able lay the groundwork to examine large-scale ecological change in a spatially quantifiable fashion. I used a subset of images from the Mountain Legacy Project to assess the accuracy and utility of the WSL Monoplotting Tool for georeferencing oblique angle photographs and measuring ecological change. I determined that the tool can georeference objects to within 15m of their real world location, and that the displacement of the geographic center of 121 control points was less than 3m away from its real world location. Most of the error in object placement was due to the angle of viewing incidence with the ground. I proposed simple rules for control point selection to reduce georeferencing errors. These rules included not placing control points on: a) surfaces that have very low angles of viewing incidence; b) on the tops of hills/ridges/terrain breaks where they could be displaced by long distances; and c) no further away than the most distance objects to be classified. I determined that this tool is versatile, accurate, and easy to learn and use.

In Chapter 4, I applied the methods developed in Chapter 3 to examine 137 image pairs from the Mountain Legacy Project collection to examine ecological change over an area of 320,000 ha in the southern Alberta Rocky Mountains and foothills region. I found that the majority of the landscape (63.4%) has remained in the same vegetation category from 1909-2008. However, of the portion that changed, the grassland-forest balance of the landscape had changed considerably over the past century in favour of forests. More than a

quarter (28%) of the landscape was in a later seral stage than it was in 1909, and less than 9% was in an earlier state. The landscape had shifted from a condition in 1909 with nearly 58% in a non-forest state (grasslands, meadows, shrubs, non-vegetated and open canopy woodlands) and 42% in closed canopy forest (conifer, broadleaf deciduous and mixedwood) to 2008 where 42% was in non-forest and 58% in forest. The change was not uniform across the entire landscape. The Montane Natural Subregion had the greatest proportion of area undergoing successional advancement, with substantial (but lesser) forward change in the Subalpine, Alpine and Foothills Parkland. With regard to grassland and open canopy woodland loss alone, 15% of the total landscape had converted from either grassland or open canopy forest to closed canopy forest. We saw that nearly 37% of historical grasslands and 80% of open canopy woodlands converted to more advanced successional types, and this appears to be due to gradual forest advancement from the historic forest edge. The causes of these changes in vegetation structure are complex, and they were related to topography and disturbance history either directly or indirectly. We are seeing an increased ecological tension between the direction of vegetation change occurring presently in favour of forest expansion, and the influence of future climate change which should instead be driving these vegetation changes in favour of grassland expansion. While I could not prove that the changes observed were directly caused by changes in the fire regime, the results are consistent with what we would expect them to be if this were the case. Further research would be required to tease apart the influences of altered fire regimes, climate change, and the loss of large grazers (bison and elk) on the landscape.

In Chapter 5, I tested the assumption held by many fire management agencies that the historic vegetation structure at the turn of the 20th century was less susceptible to burning at high intensity and over large areas than the current vegetation structure. This assumption had not been well tested previously, as we did not know enough about the historic landscape scale vegetation. I used the Burn-P3 model to compare attributes of wildfire behaviour for two different scenarios: a) the baseline scenario was the present day vegetation structure of the landscape (as of 2014); b) a historic restoration scenario where I assumed a management agency restored a landscape to its historical (1909)vegetation structure. I examined whether this historical restoration scenario resulted in changes to the overall burn probability of the landscape (either increase or decrease it), whether it would reduce the probability of high intensity wildfire (using a threshold value of 4,000 kW/m, which is considered the limit for direct ground wildfire suppression efforts), and whether it would change the size distribution of fires that occurred to more smaller fires. I used

photographs from the Mountain Legacy Project to determine what the historic vegetation composition was in the Bob Creek Wildland, which is the largest piece of intact Montane wilderness in the province of Alberta. The baseline scenario used the Government of Alberta forest fuel map from 2014. I found that the overall mean burn probability was affected very little by the historical restoration scenario, resulting in only a 1.2% reduction in the mean burn probability of the landscape. However, I did find that the pattern of burn probability changed considerably, with many areas of the landscape showing increased burn probability and others showing decreases in burn probability. The areas where the burn probability increased were associated with areas that had been changed from forest to grassland which would be expected to increase the rate of spread. The areas where the burn probability decreased were associated with portions of the landscape where the fuel changes would result in decreased rates of spread. I also found that areas with increased rates of spread had an inordinate effect on changes in burn probability, as the area with increased rate of spread was less than 15% of the landscape, whereas 50% of the landscape exhibited increased burn probability. When I only considered areas that would burn at an intensity greater than 4,000 kW/m, I found the historical restoration scenario reduced the mean burn probability of the landscape by 44%. The mean fire size was also reduced in the historical restoration scenario.

In Chapter 5 I saw only minor reductions in the overall burn probability of the landscape in the historical restoration scenario, but the spatial pattern of change reduced the likelihood of fires escaping, and there were large reductions in the likelihood of high intensity fire. By cutting the likelihood of fire burning at high intensity nearly in half, future fires would be easier to manage, and fire intensity would be considerably lower, making it much easier to protect the biological legacies in the Bob Creek Wildland. Historical restoration is not a panacea that could be applied everywhere to reduce wildfire risk, as the relative position of accelerant versus suppressant fuels is critical. In our case with the Bob Creek Wildland the fuels that decreased rates of spread fuels were downwind of the fuel changes that increased the rate of spread. The results would have likely been quite different had we not “restored” the downwind side of the wildland to broadleaf deciduous vegetation and instead left this area covered largely by coniferous forest as it is today.

6.2. Recommendations

6.2.1. *Management Recommendations*

Given the degree of vegetation change we have seen over the past century and the likelihood that these trends will continue, this landscape will continue to become more heavily forested with concomitant losses of open canopy woodlands, grasslands, and the flora and fauna adapted to these systems. Furthermore, these changes appear to have increased the probability of intense crown fires occurring on this landscape. How these changes are perceived by people and management agencies varies considerably. Some do not see these changes as problematic, whereas others see the need for action to change the future trajectory of vegetation changes and increased wildfire risk. The observed trend over the past century shows that the landscape vegetation balance between forests and grasslands is moving in favour of forests. The climate projections show that this balance should be shifting in the opposite direction. These trends will only increase the degree of mismatch between the vegetation that is actually at a particular location, and the vegetation that is most suited to that location. For this reason, it is my opinion that restoration of parts of this landscape is a reasonable and responsible thing for management agencies to consider.

The historical vegetation analysis that I used in this dissertation could be applied over a much broader geographic area, and at a finer resolution (see Research Recommendations below) to identify the areas of the landscape that have changed the most over the past century. This measure of vegetation change, in isolation, is not overly useful for guiding restoration activities, however, as it must also be coupled with an analysis of future climate suitability. Climate suitability modelling can determine areas that will best be suited to different vegetation types in the future. From these two inputs (vegetation change and climate suitability), managers could then determine the areas of the landscape that will have the least suitable vegetation on site, and these could be prioritised for ecological restoration. The Government of Alberta recently created the Castle Crown Wildland Park, and this park could be a location where these management approaches could be highly beneficial. Given that the government is embarking on creating a management plan for this new Park, I recommend giving serious consideration to ecological restoration of some areas of this new Park.

There are many ways to achieve ecological restoration goals. When restoration requires changing vegetation composition and spatial arrangement over large landscapes, many different management approaches and tools will be required. The tools and

management approaches used to achieve ecological restoration vary considerably, and different ones can be used in isolation or in combination depending on whether restoration is planned for protected areas, private lands, or publicly owned lands. These approaches include silviculture, prescribed fire, wildland fire management, and grazing management, among others. Forest management in the region could contribute to ecological restoration goals in the region and in other areas too. Timber harvesting could be directed towards reducing crown bulk density, creating more open stand structures, and in some areas grassland restoration could be achieved by complete removal of forests. These efforts would likely have to be combined with prescribed burning, grazing, and a commitment to long-term engagement in management activities and monitoring. Assisted migrations for grassland species into the Montane, and Montane species into the lower subalpine could also be considered to prepare the landscape for the future climate. Douglas-fir in particular could be planted throughout the lower elevations, and in locations where climate suitability analyses show that it will be well suited to in the future.

There is a great deal of wildland-urban interface through the Crowsnest Pass. As we saw with the 1910 fires that swept through this region and the Lost Creek fire in 2003, when the conditions are favourable, large intense and rapidly spreading crown fires are possible within this landscape. If the forest continues to encroach on the grasslands and the canopy of the forest continues to close, the communities throughout the Pass will come under greater risk of fire burning through them as we have seen occur in Slave Lake, Kelowna, Barriere, and most recently Fort McMurray. Treatment of the protected areas alone will not decrease the wildfire risk to the urban interface areas of the landscape.

Fuel treatments are often proposed as a way or reducing wildfire risks to communities and ecological values. These treatments usually focus on stand thinning to reduce crown bulk density to decrease the intensity of wildfire and reduce the probability of crown fires. However, when forests are thinned, in-stand wind speeds increase, and more light reaching the forest floor increases grass cover. Both of these increase the rate of spread of fires that occur, which can increase the wildfire risk to specific values (communities, infrastructure, key ecological sites). When conducting burn probability analyses, researchers should also consider partitioning their analyses to examine how different seasons may influence the burn probability. While the mean burn probability may decline by thinning forests, the spring burn probability may increase dramatically due to higher grassy fuel content. This does not mean that thinning is not worth doing, because this increased risk can be mitigated by deploying appropriate resources at the right locations and

times of year when the rate of spread is elevated. Furthermore, grass fires can be much more easily extinguished, but this requires firefighters to be able to reach the fire faster than the speed at which fire can spread to values at risk.

If historical restoration were to be taken seriously for the purposes of preserving ecological values and reducing wildfire risk, the location and sequence of activities is important to consider. If managers first remove large areas of forest around values at risk as fire breaks, they will end up creating more grassland, which could increase rates of spread and increase downwind burn probabilities. Rather than first creating fire breaks, it might be wise to establish fire "brakes" first, or suppressants to slow fire spread on the landscape. By increasing the cover of broadleaf deciduous vegetation cover in key locations to prevent fire spread to values at risk, the probability of burning could be reduced as well as the intensity of the fires that occur.

6.2.2. ***Research Recommendations***

1. Continued development of oblique image analysis tools and methods: When this study began, there were no user friendly, accurate methods for quantitative and spatial analysis of oblique images, but recent advancements in computing power have enabled the development of techniques for georeferencing oblique angle photographs. In addition to the WSL Monoplotting Tool (Bozzini et al. 2012) that I used in this research, in the past year two new software packages have become available (the Pic2Map plugin for QGIS (Milani and Produit 2016), and PRACTISE (Harer et al. 2016)); there are others in development that I have heard people discuss at remote sensing and cartography workshops. My analysis was decidedly crude by using a 1-ha grid size to classify images, however it was the scale at which I could balance a substantial number of images with a coarse vegetation analysis. Smaller areas can be examined at finer spatial resolution, but large landscape studies at finer resolution are time consuming. What is missing is a method of supervised image classification that can account for the varying depth of field in oblique angle images. If images could be analyzed semi-automatically at a pixel scale, far more detail could be extracted from the images, and far more area could be covered.
2. Continued examination of landscape change in the southern Rockies of Alberta: even in the absence of new tools, the methods I developed could be used to cover a much larger landscape, including the newly created Castle-Crown Wildland Park to the south

of our study area, the Kananaskis Valley, Banff National Park, the R11 Forest Management Unit, and Willmore Wilderness Park. Understanding the natural history of the landscape from 1900-present day across all of the Alberta Rockies would be invaluable. Combining image analysis with field work to collect relevant site level ecological information could help further refine our understanding of what processes have driven the changes we have observed.

3. Determination of the mechanisms driving open canopy woodland and grassland change to closed canopy forest: A finer scale analysis is required to elucidate the mechanisms driving forest encroachment. By adding a third time point to the analysis (1949 aerial photographs) we could also begin to elucidate the rate of change. Adding field work to determine the age of forests that have invaded the grasslands and filled the open canopy woodlands would help to determine the driving forces behind the advance of closed canopy forest. The age of new forests might decline with distance from the historic edge, be even aged, or have multiple cohorts. These ages could be correlated to climate factors such as changes in temperature and moisture regimes. Each of these age structure possibilities would suggest different mechanisms. If open canopy woodlands and grasslands were created and maintained by mixed severity or low severity fire, there should be a fire scar and/or soil charcoal record of this. Finding former open forests that are now closed is a difficult task, but with historical photographs, we can find exactly where there used to be open canopy forests 100 years ago and use these locations to sample.
4. Testing methods for filling gaps in visible coverage from historic photographs: Even with an extensive collection of oblique images such as the Mountain Legacy Project, some portions of the landscape are more visible than others. Sometimes an area may be visible, but the photographic quality is poor, or atmospheric condition at the time the photo was taken make it difficult to classify vegetation. I used a fairly simple method of interpolation to fill the non-visible portions of the landscape that only considered the spatial arrangement of vegetation categories. By including other variables such as elevation, aspect, insolation, site or soil moisture (i.e. Wet Areas Mapping data), more complex interpolation procedures could be used such as co-kriging and regression kriging. Using these other inputs could well improve the accuracy of interpolating what is in the non-visible portions of the landscape.
5. Burn-P3 modelling to simulate historic fire regime: There are numerous parameters that can be used to change the outputs of the Burn-P3 model. Using the fire history from

1961-2014 restricted burning in Bob Creek Wildland to only 3-4 ha per year on average. If the historical fire cycle of 15-30 years were one that a management agency wished to maintain, how many fires would need to burn, and how big would they have to be? The area within the Bob Creek Wildland that would have to burn on average every year would be 700-1400 ha. We could experiment with changing fire size class distributions, fire frequency distributions, and the duration of burning for individual fires to see if we could achieve this burn rate, and in so doing provide managers with meaningful measures for how to maintain a historical fire regime (or one of any burn rate they desired).

6.3. Final Thoughts

My interest in this research area began many years ago when I worked as the prescribed fire coordinator for the Province of Alberta. I often saw these historical photographs from the MLP collection (and others) used to show the conversion of large areas of the landscape from grasslands to forests. I began to wonder how representative these photographs being shown were of the greater landscape: were people just choosing the pictures that suited their purpose, or was there really a large scale phenomenon at play? If we were to use photographs like this to justify land management actions designed to remove the forests, or burn the forests, or restore parks as “ecologically friendly”, our decisions should be based on the best science at hand, and where we lacked that knowledge, we needed to fill the gaps. I now feel as though I have barely scratched the surface of this research area, and look forward to what new information and understanding future studies by myself and others may bring.

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Appendix A: Mountain Legacy Project Photos Used

Table A.1: List of all photographs used in Chapters 4 and 5 for analysis of landscape change. The majority of the images are from the MP Bridgland Survey of 1913-1914. Included is the year the original image was taken, and the year the repeat image was taken. Also included is the name of the photostation, and the file name used by the Mountain Legacy Project team for each image.

Surveyor	Year of Original Image	Year of Repeat Image	MLP Image Name	MLP Image Name	Photostation Name
Bridgland	1913	2007	BRI1913_B13-4-NE	MLP2007_B13-4crop	Stn. 1 Coal Creek
Bridgland	1913	2007	BRI1913_B13-7-SW	MLP2007_B13-7crop	Stn. 1 Coal Creek
Bridgland	1914	2006	BRI1914_B14-52	MLP2006_B14-52	Stn. 10 Blairmore North
Bridgland	1914	2006	BRI1914_B14-51	MLP2006_B14-51	Stn. 10 Blairmore North
Bridgland	1913	2006	BRI1913_B13-566	MLP2006_B13-566-E	Stn. 100 Vicary Divide No. 1
Bridgland	1913	2006	BRI1913_B13-567	MLP2006_B13-567-SE	Stn. 100 Vicary Divide No. 1
Bridgland	1913	2007	BRI1913_B13-569	MLP2007_B13-569	Stn. 101 Daisy North No. 1
Bridgland	1913	2007	BRI1913_B13-578-SW	MLP2007_B13-578	Stn. 103 Racehorse North No. 1
Bridgland	1913	2007	BRI1913_B13-584-W	MLP2007_B13-584	Stn. 103 Racehorse North No. 1
Bridgland	1913	2007	BRI1913_B13-587-SE	MLP2007_B13-587	Stn. 104 Racehorse North No. 2
Bridgland	1913	2006	BRI1913_B13-601	MLP2006_B13-601-NW	Stn. 106 Racehorse South No. 2
Bridgland	1913	2006	BRI1913_B13-611	MLP2006_B13-611-NE	Stn. 108 Allison Peak No. 2
Bridgland	1913	2007	BRI1913_B13-615-SE-crop	MLP2007_B13-615-crop	Stn. 109 Racehorse Overlook
Bridgland	1913	2007	BRI1913_B13-72-NE	MLP2007_B13-72	Stn. 11 Boundary No. 1
Bridgland	1913	2007	BRI1913_B13-74-E	MLP2007_B13-74	Stn. 11 Boundary No. 1
Bridgland	1913	2007	BRI1913_B13-73-NE	MLP2007_B13-73	Stn. 11 Boundary No. 1
Bridgland	1913	2007	BRI1913_B13-593-NE	MLP2007_B13-593	Stn. 110 Daisy West No. 3
Bridgland	1913	2007	BRI1913_B13-596-W	MLP2007_B13-596	Stn. 110 Daisy West No. 3
Bridgland	1913	2006	BRI1913_B13-636	MLP2006_B13-636-E	Stn. 112 Gold Creek West No. 2
Bridgland	1913	2006	BRI1913_B13-638	MLP2006_B13-638-S	Stn. 113 & 114 Lille North No. 1
Bridgland	1913	2006	BRI1913_B13-640	MLP2006_B13-640-W	Stn. 113 & 114 Lille North No. 1
Bridgland	1913	2006	BRI1913_B13-642	MLP2006_B13-642-W	Stn. 113 & 114 Lille North No. 1
Bridgland	1913	2006	BRI1913_B13-643	MLP2006_B13-643-NW	Stn. 113 & 114 Lille North No. 1
Bridgland	1913	2006	BRI1913_B13-649	MLP2006_B13-649	Stn. 115 Turtle Mt. No. 1
Bridgland	1913	2006	BRI1913_B13-653	MLP2006_B13-653	Stn. 116 Turtle Mt. No. 2
Bridgland	1913	2006	BRI1913_B13-658	MLP2006_B13-658	Stn. 117 Coleman North
Bridgland	1913	2006	BRI1913_B13-660	MLP2006_B13-660	Stn. 117 Coleman North
Bridgland	1913	2006	BRI1913_B13-665	MLP2006_B13-665	Stn. 118 Mt. Phillips
Bridgland	1913	2006	BRI1913_B13-666	MLP2006_B13-666	Stn. 118 Mt. Phillips
Bridgland	1913	2006	BRI1913_B13-668	MLP2006_B13-668	Stn. 118 Mt. Phillips

Bridgland	1913	2006	BRI1913_B13-667	MLP2006_B13-667	Stn. 118 Mt. Phillips
Bridgland	1913	2006	BRI1913_B13-677	MLP2006_B13-677-NW	Stn. 119 Willoughby Ridge No. 2
Bridgland	1913	2006	BRI1913_B13-678	MLP2006_B13-678-E	Stn. 119 Willoughby Ridge No. 2
Bridgland	1913	2006	BRI1913_B13-671	MLP2006_B13-671-NE	Stn. 119 Willoughby Ridge No. 2
Bridgland	1913	2008	BRI1913_B13-82	MLP2008_B13-82	Stn. 12 Boundary No. 2A
Bridgland	1913	2006	BRI1913_B13-682	MLP2006_B13-682	Stn. 121 Near Mt. Coleman
Bridgland	1913	2006	BRI1913_B13-689	MLP2006_B13-689	Stn. 122 Hillcrest Mt.
Bridgland	1913	2007	BRI1913_B13-95-NE	MLP2007_B13-95	Stn. 14 Livingstone West No. 1
Bridgland	1913	2007	BRI1913_B13-100-E	MLP2007_B13-100	Stn. 14 Livingstone West No. 1
Bridgland	1913	2007	BRI1913_B13-107-E	MLP2007_B13-107	Stn. 17 Forest Divide No. 1
Bridgland	1913	2007	BRI1913_B13-112-S	MLP2007_B13-112	Stn. 18 Pasque Mt. No. 2
Bridgland	1913	2007	BRI1913_B13-17-NE	MLP2007_B13-17	Stn. 2 Livingstone East No. 1
Bridgland	1913	2007	BRI1913_B13-13-NW	MLP2007_B13-13	Stn. 2 Livingstone East No. 1
Bridgland	1913	2007	BRI1913_B13-119-NW	MLP2007_B13-119	Stn. 20 North Twin No. 2
Bridgland	1913	2007	BRI1913_B13-128-E	MLP2007_B13-128	Stn. 22 South Twin
Bridgland	1913	2008	BRI1913_B13-134	MLP2008_B13-134	Stn. 23 Cabin Ridge No. 1
Bridgland	1913	2007	BRI1913_B13-155-S-crop	MLP2007_B13-155-crop	Stn. 26 Fly Hill
Bridgland	1913	2008	BRI1913_B13-157	MLP2008_B13-157	Stn. 27 Dutch Creek Head No. 1
Bridgland	1913	2008	BRI1913_B13-163	MLP2008_B13-163	Stn. 29 Hidden Creek No. 1
Bridgland	1914	2006	BRI1914_B14-188	MLP2006_B14-188	Stn. 29 Passburg North
Bridgland	1914	2006	BRI1914_B14-194	MLP2006_B14-194	Stn. 30 Dipper
Bridgland	1913	2008	BRI1913_B13-177	MLP2008_B13-177	Stn. 31 Dutch Creek North No. 2
Bridgland	1913	2008	BRI1913_B13-175	MLP2008_B13-175	Stn. 31 Dutch Creek North No. 2
Bridgland	1913	2008	BRI1913_B13-179	MLP2008_B13-179_a	Stn. 32 Daisy North No. 3
Bridgland	1913	2007	BRI1913_B13-186-N	MLP2007_B13-186	Stn. 33 Horseshoe Ridge
Bridgland	1913	2007	BRI1913_B13-189-E	MLP2007_B13-189	Stn. 33 Horseshoe Ridge
Bridgland	1913	2007	BRI1913_B13-190	MLP2007_B13-190	Stn. 34 Isolated Peak No. 1
Bridgland	1913	2007	BRI1913_B13-203-W	MLP2007_B13-203	Stn. 36 Mt. Livingstone
Bridgland	1913	2007	BRI1913_B13-222-NW	MLP2007_B13-222	Stn. 39 Thunder Mountain No. 2
Bridgland	1913	2007	BRI1913_B13-223-NW	MLP2007_B13-223	Stn. 39 Thunder Mountain No. 2
Bridgland	1913	2007	BRI1913_B13-256-S	MLP2007_B13-256	Stn. 40 Thunder Mountain No. 1
Bridgland	1913	2007	BRI1913_B13-246-E	MLP2007_B13-246	Stn. 40 Thunder Mountain No. 1
Bridgland	1913	2007	BRI1913_B13-245-E	MLP2007_B13-245	Stn. 40 Thunder Mountain No. 1
Bridgland	1913	2008	BRI1913_B13-258-SE	MLP2008_B13-258-SE	Stn. 43 Grassy Ridge
Bridgland	1913	2008	BRI1913_B13-260-SW	MLP2008_B13-260-SW	Stn. 43 Grassy Ridge
Bridgland	1913	2008	BRI1913_B13-261-W	MLP2008_B13-261-W	Stn. 43 Grassy Ridge
Bridgland	1913	2007	BRI1913_B13-267-SE	MLP2007_B13-267	Stn. 44 Riley Creek No. 1
Bridgland	1913	2007	BRI1913_B13-270-N	MLP2007_B13-270	Stn. 45 Riley Creek No. 3
Bridgland	1913	2007	BRI1913_B13-275-SW	MLP2007_B13-275	Stn. 46 Riley Creek No. 2

Bridgland	1913	2007	BRI1913_B13-277-SW	MLP2007_B13-277	Stn. 47 Livingstone Centre No. 2
Bridgland	1913	2007	BRI1913_B13-280-SE	MLP2007_B13-280	Stn. 47 Livingstone Centre No. 2
Bridgland	1913	2007	BRI1913_B13-284-SW	MLP2007_B13-284	Stn. 48 Livingstone Centre No. 1
Bridgland	1913	2007	BRI1913_B13-301-S	MLP2007_B13-301	Stn. 51 Plateau Mt. No. 3
Bridgland	1913	2007	BRI1913_B13-308-E	MLP2007_B13-308	Stn. 53 Sentinel Pass West No. 1
Bridgland	1913	2007	BRI1913_B13-309-SE	MLP2007_B13-309	Stn. 53 Sentinel Pass West No. 1
Bridgland	1913	2007	BRI1913_B13-315-NW	MLP2007_B13-315	Stn. 55 Hailstone Butte No. 1
Bridgland	1913	2007	BRI1913_B13-319-SE	MLP2007_B13-319	Stn. 56 Hailstone Butte No. 3
Bridgland	1913	2008	BRI1913_B13-325-W	MLP2008_B13-325_W_a	Stn. 57 Willow Creek No. 1
Bridgland	1913	2008	BRI1913_B13-328-SE	MLP2008_B13-328_SE_a	Stn. 57 Willow Creek No. 1
Bridgland	1913	2008	BRI1913_B13-42	MLP2008_B13-42	Stn. 6 Bolton No. 1
Bridgland	1913	2008	BRI1913_B13-45	MLP2008_B13-45	Stn. 6 Bolton No. 1
Bridgland	1913	2007	BRI1913_B13-340-N	MLP2007_B13-340	Stn. 60 Windy Peak No. 1
Bridgland	1913	2007	BRI1913_B13-352-SE	MLP2007_B13-352	Stn. 62 Bruin No. 1
Bridgland	1913	2008	BRI1913_B13-355-NW	MLP2008_B13-355	Stn. 63 Cervus No. 1
Bridgland	1913	2007	BRI1913_B13-367-E	MLP2007_B13-367	Stn. 65 Bruin No. 2
Bridgland	1913	2007	BRI1913_B13-375-E	MLP2007_B13-375	Stn. 66 Camp Creek No. 1
Bridgland	1913	2007	BRI1913_B13-374-NE	MLP2007_B13-374	Stn. 66 Camp Creek No. 1
Bridgland	1913	2007	BRI1913_B13-376-SE	MLP2007_B13-376	Stn. 66 Camp Creek No. 1
Bridgland	1913	2007	BRI1913_B13-378-SE	MLP2007_B13-378	Stn. 67 Camp Creek No. 2
Bridgland	1913	2007	BRI1913_B13-381-NE	MLP2007_B13-381	Stn. 67 Camp Creek No. 2
Bridgland	1913	2007	BRI1913_B13-379-E	MLP2007_B13-379	Stn. 67 Camp Creek No. 2
Bridgland	1913	2008	BRI1913_B13-385	MLP2008_B13-385	Stn. 69 sem. 6-13-2-5
Bridgland	1913	2008	BRI1913_B13-386	MLP2008_B13-386	Stn. 69 sem. 6-13-2-5
Bridgland	1913	2008	BRI1913_B13-50	MLP2008_B13-50	Stn. 7 Bolton No. 2
Bridgland	1913	2007	BRI1913_B13-387-SW	MLP2007_B13-387	Stn. 70 Chaffen
Bridgland	1913	2007	BRI1913_B13-389-NW	MLP2007_B13-389	Stn. 70 Chaffen
Bridgland	1913	2007	BRI1913_B13-390-N	MLP2007_B13-390	Stn. 70 Chaffen
Bridgland	1913	2007	BRI1913_B13-391-E	MLP2007_B13-391	Stn. 70 Chaffen
Bridgland	1913	2007	BRI1913_B13-393-S	MLP2007_B13-393	Stn. 70 Chaffen
Bridgland	1913	2008	BRI1913_B13-396	MLP2008_B13-396	Stn. 71 White Creek Head
Bridgland	1913	2008	BRI1913_B13-397	MLP2008_B13-397	Stn. 71 White Creek Head
Bridgland	1913	2008	BRI1913_B13-400	MLP2008_B13-400_a	Stn. 72 Chimney Ridge
Bridgland	1913	2008	BRI1913_B13-402	MLP2008_B13-402	Stn. 72 Chimney Ridge
Bridgland	1913	2008	BRI1913_B13-399	MLP2008_B13-399_a	Stn. 72 Chimney Ridge
Bridgland	1913	2008	BRI1913_B13-403	MLP2008_B13-403	Stn. 72 Chimney Ridge
Bridgland	1913	2008	BRI1913_B13-405	MLP2008_B13-405	Stn. 73 White Creek East
Bridgland	1913	2008	BRI1913_B13-407	MLP2008_B13-407	Stn. 73 White Creek East
Bridgland	1913	2008	BRI1913_B13-408	MLP2008_B13-408	Stn. 73 White Creek East
Bridgland	1913	2008	BRI1913_B13-409	MLP2008_B13-409	Stn. 73 White Creek East
Bridgland	1913	2008	BRI1913_B13-410	MLP2008_B13-410	Stn. 73 White Creek East

Bridgland	1913	2007	BRI1913_B13-415-N-crop	MLP2007_B13-415-crop	Stn. 74 Ernst Creek No. 1
Bridgland	1913	2007	BRI1913_B13-416-W-crop	MLP2007_B13-416-crop	Stn. 74 Ernst Creek No. 1
Bridgland	1913	2007	BRI1913_B13-418-S-crop	MLP2007_B13-418-crop	Stn. 74 Ernst Creek No. 1
Bridgland	1913	2007	BRI1913_B13-426-NE	MLP2007_B13-426	Stn. 76 Vicary West No. 1
Bridgland	1913	2007	BRI1913_B13-428-E	MLP2007_B13-428	Stn. 76 Vicary West No. 1
Bridgland	1913	2007	BRI1913_B13-431-E	MLP2007_B13-431	Stn. 77 Vicary West No. 2
Bridgland	1913	2008	BRI1913_B13-452-E	MLP2008_B13-452-E	Stn. 81 Elk River No. 1
Bridgland	1913	2007	BRI1913_B13-454-S-crop	MLP2007_B13-454-crop	Stn. 82 Daisy East No. 1
Bridgland	1913	2006	BRI1913_B13-466	MLP2006_B13-466-SW	Stn. 84 Centre Peak
Bridgland	1913	2006	BRI1913_B13-469	MLP2006_B13-469-NE	Stn. 84 Centre Peak
Bridgland	1913	2006	BRI1913_B13-470	MLP2006_B13-470-E	Stn. 84 Centre Peak
Bridgland	1913	2006	BRI1913_B13-485	MLP2006_B13-485-W	Stn. 86 Lille East
Bridgland	1913	2006	BRI1913_B13-487	MLP2006_B13-487-N	Stn. 86 Lille East
Bridgland	1913	2006	BRI1913_B13-490	MLP2006_B13-490-NE	Stn. 87 Prairie Overlook
Bridgland	1913	2006	BRI1913_B13-492	MLP2006_B13-492-SE	Stn. 87 Prairie Overlook
Bridgland	1913	2006	BRI1913_B13-491	MLP2006_B13-491-E	Stn. 87 Prairie Overlook
Bridgland	1913	2006	BRI1913_B13-495	MLP2006_B13-495-NW	Stn. 88 Willoughby North
Bridgland	1913	2006	BRI1913_B13-496	MLP2006_B13-496-N	Stn. 88 Willoughby North
Bridgland	1913	2006	BRI1913_B13-500	MLP2006_B13-500	Stn. 89 Sentry Mt.
Bridgland	1913	2006	BRI1913_B13-501	MLP2006_B13-501	Stn. 89 Sentry Mt.
Bridgland	1913	2007	BRI1913_B13-57-W	MLP2007_B13-57	Stn. 9 Cyclamen Ridge No. 1
Bridgland	1913	2006	BRI1913_B13-520	MLP2006_B13-520-SW	Stn. 93 Cow Creek No. 1
Bridgland	1913	2006	BRI1913_B13-521	MLP2006_B13-521-W	Stn. 93 Cow Creek No. 1
Bridgland	1913	2006	BRI1913_B13-524	MLP2006_B13-524-SW	Stn. 94 Cow Creek No. 2
Bridgland	1913	2007	BRI1913_B13-542-NE-crop	MLP2007_B13-542-crop	Stn. 97 Vicary North
Bridgland	1913	2007	BRI1913_B13-551-W-crop	MLP2007_B13-551-crop	Stn. 98 Daisy West No. 1
Bridgland	1913	2006	BRI1913_B13-555	MLP2006_B13-555-NW	Stn. 99 Ma Butte
Bridgland	1913	2006	BRI1913_B13-556	MLP2006_B13-556-N	Stn. 99 Ma Butte
Bridgland	1913	2006	BRI1913_B13-554	MLP2006_B13-554-SW	Stn. 99 Ma Butte
Sheppard	1914		6	No Repeat	Whaleback
Sheppard	1914		7	No Repeat	Whaleback

Appendix B: Vegetation Classification Examples from Mountain Legacy Images used in Chapters 4 and 5

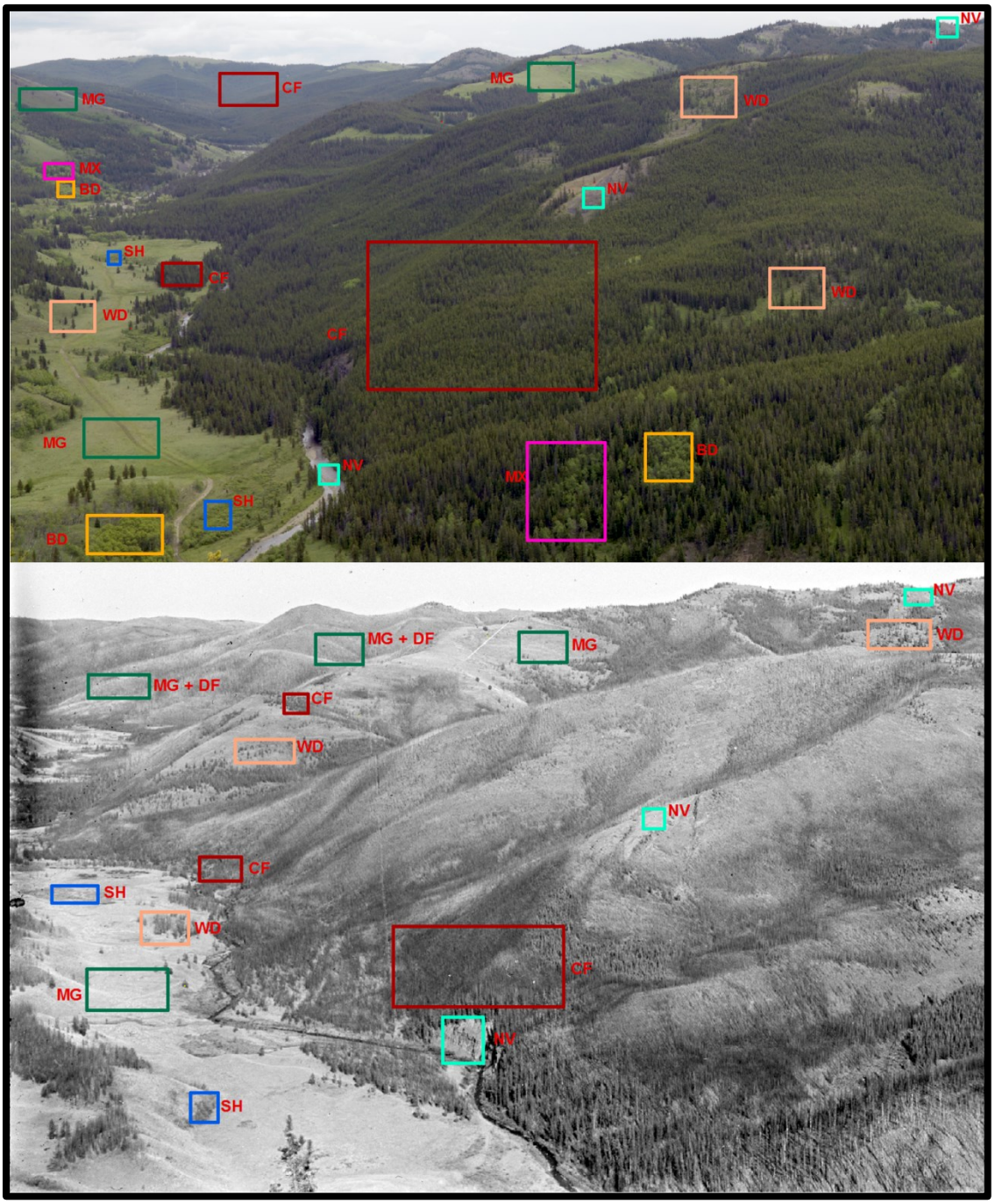

Figure B.1: Vegetation class examples from image pair 0074 (BRI1913_B13-352-SE and MLP2007_B13-352). CF = conifer, BD = broadleaf deciduous, MX = mixedwood, WD = open canopy woodland, MG = meadow/grassland, SH = shrubland, NV = non-vegetated, DF = disturbed fire. See Table 4.2 and Section 4.3.2 for further description of photographic signatures used to classify.

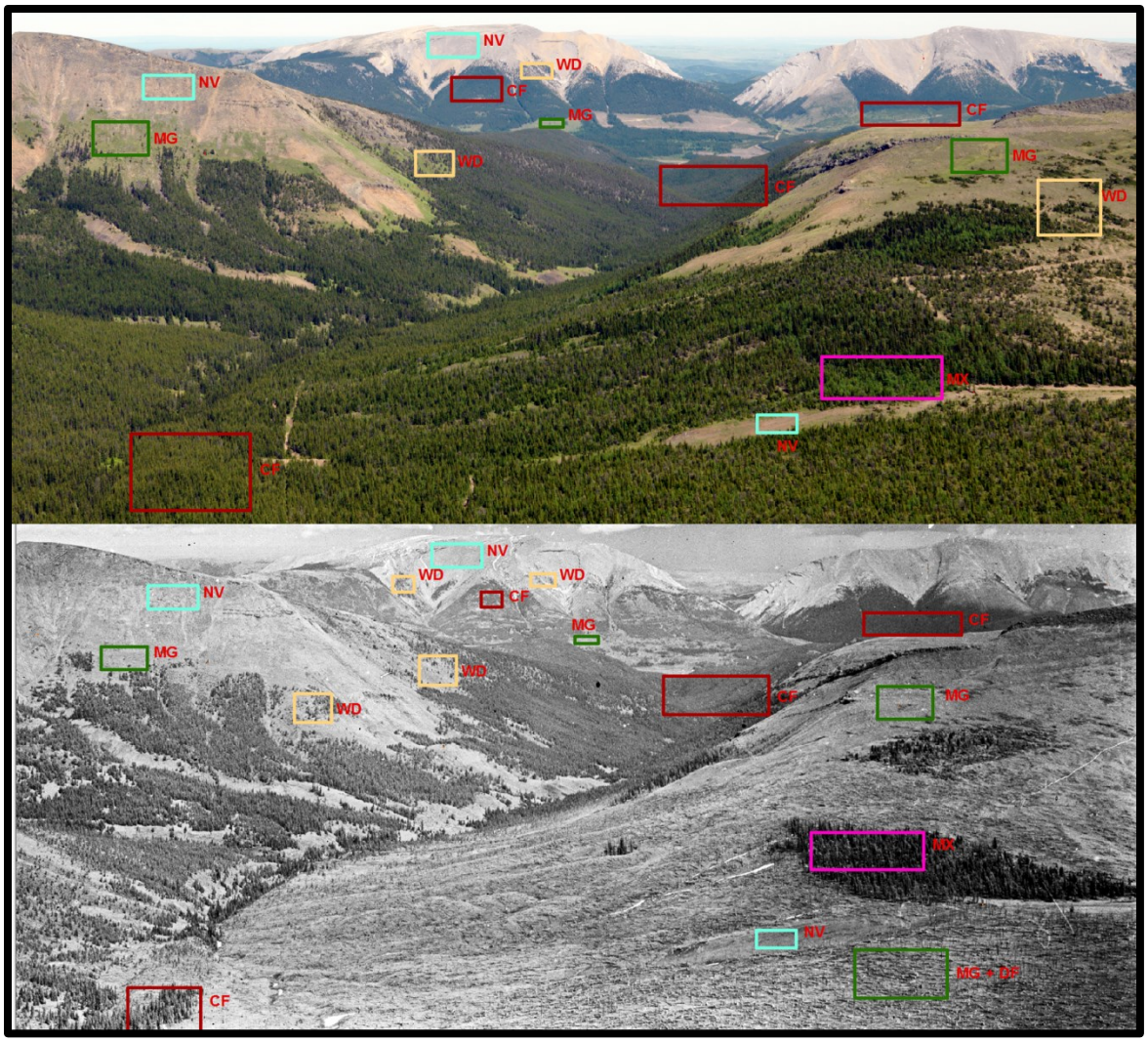

Figure B.2: Vegetation class examples from image pair 1213 (BRI1913_B13-100-E and MLP2007_B13-100). CF = conifer, BD = broadleaf deciduous, MX = mixedwood, WD = open canopy woodland, MG = meadow/grassland, SH = shrubland, NV = non-vegetated, DF = disturbed fire. See Tables 4.2 and Section 4.3.2 for further description of photographic signatures used to classify.

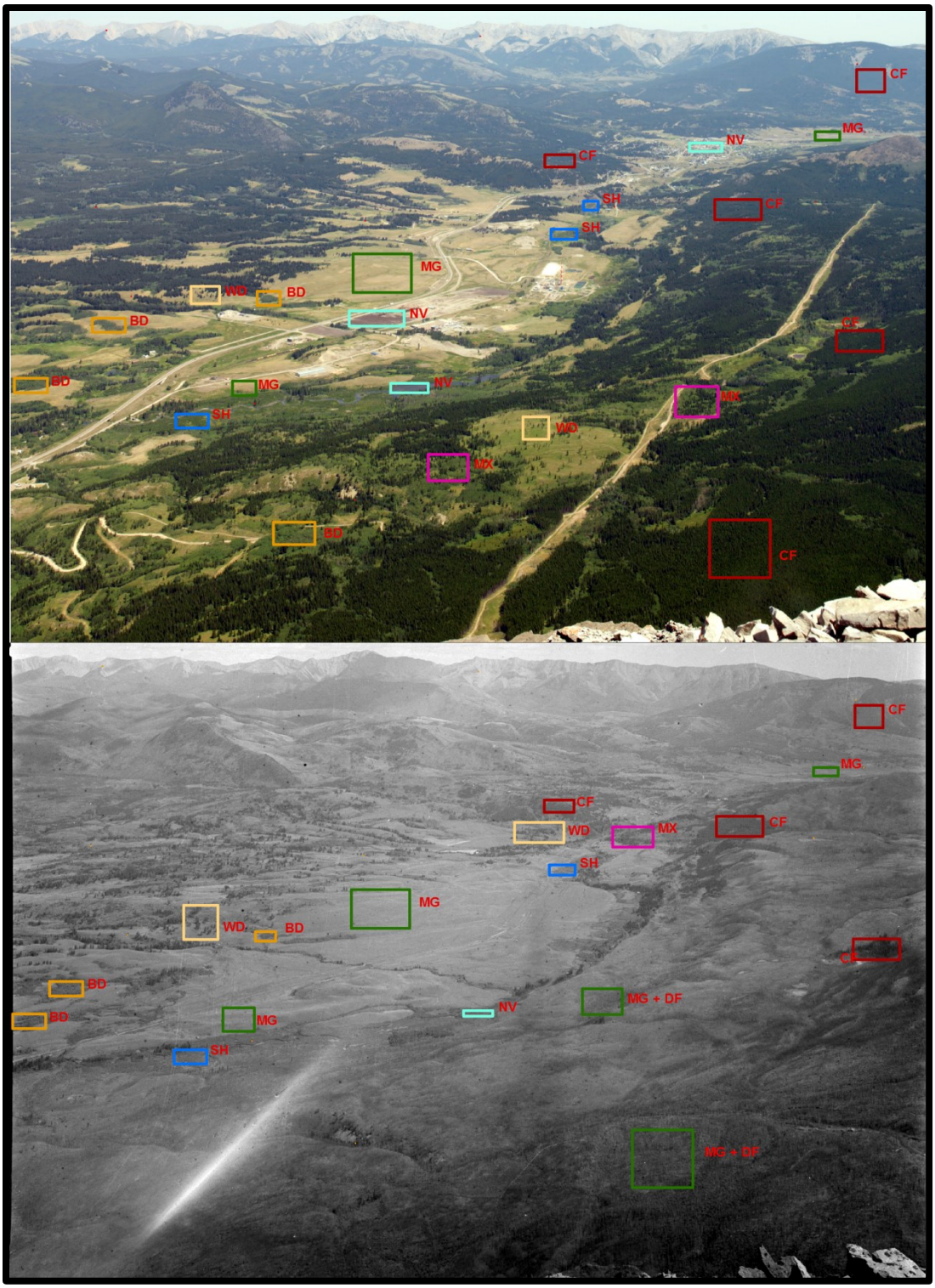

Figure B.3: Vegetation class examples from image pair 0145 (BRI1913_B13-501 and MLP2007_B13-501). CF = conifer, BD = broadleaf deciduous, MX = mixedwood, WD = open canopy woodland, MG = meadow/grassland, SH = shrubland, NV = non-vegetated, DF = disturbed fire. See Table 4.2 and Section 4.3.2 for further description of photographic signatures used to classify.

Appendix C: Statistical Outputs From Chapters 4 and 5

Chapter 4: Landscape Change

The ordinal logistic regression statistical analyses conducted in Chapter 4 are summarized below. The outputs presented here relate to the hypotheses described in the introduction of Chapter 4. All of these statistical tests are summarized in the methods section of Chapter 4 of this dissertation. The outputs of the various statistical tests are all shown below.

Analysis 1: Landscape vegetation change

The first analysis conducted was to determine whether the landscape level change in the successional state of the vegetation observed over 99 years was due to topographic factors (elevation and solar radiation) and previous known disturbances such as fire, harvesting or human land use. Table C.1 below shows all the models that were tested to determine which one best explained the whether or not vegetation had succeeded to a more advanced successional state, remained the same, or move backward. The statistical methods used are described in Chapter 4, section 4.3.5.1.

Table C.1: Summary of Akaike Information Criterion (AIC), ∆AIC and AIC weights (AICw) for all possible ordinal regression models on succession change (see Table 4.8) built using the variables of solar radiation (Z.solar), time since harvest (TSH), time since fire (TSF), anthropogenic disturbance (DA), elevation (Z.elev), and the random block effect of watersheds (blockshed). The models are sorted by lowest AIC value to highest.

Model Variables	AIC	∆AIC	AIC*w*
Z.solar, TSH, TSF, DA, Z.elev, Blockshed	276695.97	0	1
Z.solar, TSH, TSF, Z.elev, Blockshed	276730	34.03	<.000001
TSH, TSF, DA, Z.elev, Blockshed	276774	78.03	<.000001
TSH, TSF, Z.elev, Blockshed	276785.69	89.72	<.000001
Z.solar, TSH, TSF, DA, Blockshed	277239.91	543.94	<.000001
TSH, TSF, DA, Blockshed	277324.14	628.17	<.000001
Z.solar, TSH, TSF, Blockshed	277444	748.03	<.000001
TSH, TSF, Blockshed	277558.18	862.21	<.000001
Z.solar, TSH, DA, Z.elev, Blockshed	283915	7219.03	0
Z.solar, TSH, Z.elev, Blockshed	283939.84	7243.87	0
TSH, DA, Z.elev, Blockshed	283961.74	7265.77	0
TSH, DA, Blockshed	283965.42	7269.45	0
TSH, Z.elev, Blockshed	283991	7295.03	0
Z.solar, TSH, DA, Blockshed	284583.05	7887.08	0
Z.solar, TSH, Blockshed	284784.32	8088.35	0
Z.solar, TSF, DA, Z.elev, Blockshed	285424	8728.03	0
TSF, DA, Z.elev, Blockshed	285548.95	8852.98	0
Z.solar, TSF, Z.elev, Blockshed	285675.03	8979.06	0
TSF, Z.elev, Blockshed	285826.98	9131.01	0
Z.solar, TSF, DA, Blockshed	286441.14	9745.17	0
TSF, DA, Blockshed	287203.59	10507.62	0
Z.solar, TSF, Blockshed	287245.15	10549.18	0
TSH, Blockshed	287312.4	10616.43	0
TSF, Blockshed	287555.91	10859.94	0
Z.solar, DA, Z.elev, Blockshed	292211.34	15515.37	0
DA, Z.elev, Blockshed	292327.16	15631.19	0
Z.solar, Z.elev, Blockshed	292426.96	15730.99	0
Z.elev, Blockshed	292566.39	15870.42	0
Z.solar, TSF, Blockshed	293387.5	16691.53	0
DA, Blockshed	293584.92	16888.95	0
Z.solar, Blockshed	294175.34	17479.37	0
Blockshed	294470	17774.03	0

From Table C.1 above, the best model was the one represented by the lowest AIC score. Following are the raw outputs from the CLMM Procedure in R. The variable “posneg” is an abbreviation for “positive or negative successional change”. The model was described as (see Table C.1 for variable abbreviation definitions):

```
formula: posneg ~ Z.solar + tsh + tsf + DA + Z.elev + (1 | blockshed)
data:    vegtrans
```

The following describes model parameters and are the defaults associated with the CLMM procedure. “Link” is the link function, “threshold” specifies the potential structure for thresholds, “nobs” is the number of observations, “logLik” is the value of the log likelihood function, “AIC” is the Akaike Information Criterion value, “niter” is the number of Newton iterations, “max.grad” is the vector of gradients for the coefficients at the estimated optimum, and “cond.H” is the condition number of the Hessian matrix at the optimum.

link	threshold	nobs	logLik	AIC	niter	max.grad	cond.H
logit	flexible	182583	-138334.98	276695.97	1963(10037)	2.99e-01	1.5e+03

The following is the output for this model described above, and under the default settings for the CLMM procedure.

```
Random effects:
Groups     Name        Variance   Std.Dev.
blockshed  (Intercept) 0.757      0.8701
Number of groups:  blockshed 58

Coefficients:
          Estimate     Std. Error   z value    Pr(>|z|)
Z.solar   -0.036420    0.005102     -7.139     9.40e-13 ***
tshS      0.396770     0.042746     9.282      < 2e-16 ***
tshM      1.707110     0.046989     36.330     < 2e-16 ***
tshL      1.610271     0.059468     27.078     < 2e-16 ***
tshN      2.489232     0.032069     77.622     < 2e-16 ***
tsfM      1.841968     0.061512     29.945     < 2e-16 ***
tsfL      0.243333     0.060484     4.023      5.74e-05 ***
tsfXL     2.150354     0.058270     36.903     < 2e-16 ***
DA        -0.068751    0.011330     -6.068     1.29e-09 ***
Z.elev    0.208548     0.008931     23.351     < 2e-16 ***
---
   Signif. codes:  0 ‘***’ 0.001 ‘**’ 0.01 ‘*’ 0.05 ‘.’ 0.1 ‘ ’ 1

Threshold coefficients:
        Estimate Std. Error  z value
 -1|0   1.4377   0.1298      11.07
  0|1   5.3604   0.1308      40.98
```

The Spearman Rank Correlations were computed as described in section 4.3.5.1 to determine if there were any strong correlations between the predictor variables. These statistics do not have associated p-values because most of the input variables are categorical.

Table C.2: Spearman's Rho statistic for correlations between all variables in the best model for landscape vegetation change. Variables are: Solar radiation (solar), time since harvest (TSH), time since fire (TSF), anthropogenic disturbance (DA), elevation (elevation.

Variable 1	Variable 2	Rho
TSF	TSH	-0.0226
TSF	DA	0.0297
TSF	Solar	0.0237
TSF	Elevation	-0.1295
TSH	DA	-0.1705
TSH	Solar	-0.1256
TSH	Elevation	-0.1707
DA	Solar	0.0240
DA	Elevation	-0.3184
Solar	Elevation	0.1786

Analysis 2: Grassland/Meadow vegetation change

The second analysis conducted was to determine whether the degree of successional change observed over 99 years in portions of the landscape that were historically in a grassland/meadow condition was due to topographic factors (elevation and solar radiation) and previous known disturbances such as fire, or human land use. Table C.3 below shows all the models that were tested to determine which one best explained the the degree of successional change. The response variable was the degree of vegetation change, coded as to indicate reverse succession (-1), no change (0), and forward change (values of 1 to 3 to indicate the degree of change, see Table 4.3 for full description of these changes). The statistical methods used are described in Chapter 4, section 4.3.5.2.

Table C.3: Summary of Akaike Information Criterion (AIC) , ∆AIC and AIC weights (AICw) for all possible ordinal regression models on the magnitude and direction of vegetation change in former grasslands (see Table 4.3) built using the variables of solar radiation (Z.solar), time since fire (TSF), anthropogenic disturbance (DA), elevation (Z.elev), and the random block effect of watersheds (blockshed). The models are sorted by lowest AIC value to highest.

Model Variables	AIC	∆AIC	AIC*w*
Z.solar, TSF, Z.elev, Blockshed	106887	0	.62246
Z.solar, TSF, DA, Z.elev, Blockshed	106888	1	.37754
Z.solar, Z.elev, Blockshed	107197	310	<.00001
Z.solar, DA, Z.elev, Blockshed	107199	312	<.00001
TSF, DA, Z.elev, Blockshed	110484	3597	0
TSF, Z.elev, Blockshed	110486	3599	0
DA, Z.elev, Blockshed	110691	3804	0
Z.elev, Blockshed	110695	3808	0
Z.solar, TSF, DA, Blockshed	110944	4057	0
Z.solar, TSF, Blockshed	111256	4369	0
Z.solar, DA, Blockshed	111318	4431	0
Z.solar	111685	4798	0
TSF, DA, Blockshed	113979	7092	0
DA, Blockshed	114176	7289	0
TSF, Blockshed	114361	7474	0
Blockshed	114585	7698	0

From Table C.3 above, the best model was the one represented by the lowest AIC score. Following are the raw outputs from the CLMM Procedure in R. The variable “chgdir” is an abbreviation for “change direction and magnitude” and is the response variable for the degree of vegetation change. The model was described as (see Table C.3 for variable abbreviation definitions):

```
formula: chgdir ~ Z.solar + tsf + Z.elev + (1 | blockshed)
data:    MG09
```

The following describes model parameters and are the defaults associated with the CLMM procedure. “Link” is the link function, “threshold” specifies the potential structure for thresholds, “nobs” is the number of observations, “logLik” is the value of the log likelihood function, “AIC” is the Akaike Information Criterion value, “niter” is the number of Newton iterations, “max.grad” is the vector of gradients for the coefficients at the estimated optimum, and “cond.H” is the condition number of the Hessian matrix at the optimum.

Link	threshold	nobs	logLik	AIC	niter	max.grad	cond.H
Logit	flexible	75696	-53435.65	106887.30	740(5390)	4.79e-03	8.2e+03

The following is the output for this model described above, and under the default settings for the CLMM procedure.

```
Random effects:
Groups      Name        Variance Std.Dev.
blockshed   (Intercept) 2.542    1.594
   Number of groups:  blockshed 57

Coefficients:
            Estimate    Std. Error    z value    Pr(>|z|)
Z.solar     -0.545423   0.009655      -56.49     <2e-16 ***
tsfM        1.602455    0.099839      16.05      <2e-16 ***
tsfXL       1.581666    0.093459      16.92      <2e-16 ***
Z.elev      1.073526    0.016825       63.80     <2e-16 ***
---
   Signif. codes:  0 ‘***’ 0.001 ‘**’ 0.01 ‘*’ 0.05 ‘.’ 0.1 ‘ ’ 1

Threshold coefficients:
        Estimate   Std. Error   z value
0|1     1.8594     0.2339       7.949
1|2     2.3318     0.2340       9.965
2|3     3.1546     0.2342       13.470
```

The Spearman Rank Correlations were computed as described in section 4.3.5.2 to determine if there were any strong correlations between the predictor variables. These statistics do not have associated p-values because most of the input variables are categorical.

Table C.4: Spearman's Rho statistic for correlations between variables in the best model for meadow and grassland vegetation change. Variables are: Solar radiation (solar), time since fire (TSF), elevation (elevation).

Variable 1	Variable 2	Rho
TSF	Solar	-0.0325
TSF	Elevation	-0.2283
Solar	Elevation	0.2351

Analysis 3: Open Canopy Woodland vegetation change

The third analysis conducted was to determine whether the degree of successional change observed over 99 years in portions of the landscape that were historically in a open canopy woodland condition was due to topographic factors (elevation and solar radiation) and previous known disturbances such as fire, or human land use. Table C.5 below shows all the models that were tested to determine which one best explained the the degree of successional change. The response variable was the degree of vegetation change, coded as to indicate reverse succession (-1), no change (0), and forward change (values of 1 to 3 to indicate the degree of change, see Table 4.3 for full description of these changes). The statistical methods used are described in Chapter 4, section 4.3.5.2.

Table C.5: Summary of Akaike Information Criterion (AIC) , ∆AIC and AIC weights (AICw) for all possible ordinal regression models on the magnitude and direction of vegetation change in former open canopy woodlands (see Table 4.3) built using the variables of solar radiation (Z.solar), time since fire (TSF), anthropogenic disturbance (DA), elevation (Z.elev), and the random block effect of watersheds (blockshed). The models are sorted by lowest AIC value to highest.

Model Variables	AIC	∆AIC	AIC*w*
Z.solar, TSF, Z.elev, Blockshed	20699	0	.7298
Z.solar, TSF, DA, Z.elev, Blockshed	20701	2	.2685
Z.solar, TSF, DA, Blockshed	20712	13	.0011
Z.solar, TSF, Blockshed	20713	14	.0001
TSF, Z.elev, Blockshed	20743	44	<.0001
TSF, DA, Z.elev, Blockshed	20745	46	<.0001
TSF, DA, Blockshed	20758	59	<.0001
TSF, Blockshed	20759	60	<.0001
Z.solar, Z.elev, Blockshed	21107	408	<.0001
Z.solar, DA, Z.elev, Blockshed	21109	410	<.0001
Z.solar, DA, Blockshed	21141	442	<.0001
Z.solar, Blockshed	21147	448	<.0001
Z.elev, Blockshed	21148	449	<.0001
DA, Z.elev, Blockshed	21150	451	<.0001
DA, Blockshed	21186	487	<.0001
Blockshed	21192	493	<.0001

From Table C.5 above, the best model was the one represented by the lowest AIC score. Following are the raw outputs from the CLMM Procedure in R. The variable “chgdir” is an abbreviation for “change direction and magnitude” and is the response variable for the degree of vegetation change. The model was described as (see Table C.5 for variable abbreviation definitions):

```
formula: chgdir ~ Z.solar + tsf + Z.elev + (1 | blockshed)
data:    WD09
```

The following describes model parameters and are the defaults associated with the CLMM procedure. “Link” is the link function, “threshold” specifies the potential structure for thresholds, “nobs” is the number of observations, “logLik” is the value of the log likelihood function, “AIC” is the Akaike Information Criterion value, “niter” is the number of Newton iterations, “max.grad” is the vector of gradients for the coefficients at the estimated optimum, and “cond.H” is the condition number of the Hessian matrix at the optimum.

link	threshold	nobs	logLik	AIC	niter	max.grad	cond.H
logit	flexible	12363	-10341.91	20699.82	743(4531)	3.18e-03	1.5e+03

The following is the output for this model described above, and under the default settings for the CLMM procedure.

```
Random effects:
Groups      Name        Variance Std.Dev.
blockshed   (Intercept) 2.41     1.553
   Number of groups:  blockshed 56

Coefficients:
             Estimate    Std. Error   z value     Pr(>|z|)
Z.solar      -0.14716    0.02201      -6.686      2.29e-11 ***
tsf[T.M]     3.65383     0.29373      12.440      < 2e-16 ***
tsf[T.XL]    4.32034     0.28494      15.162      < 2e-16 ***
Z.elev       -0.14641    0.03669      -3.991      6.58e-05 ***
---
   Signif. codes:  0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1

Threshold coefficients:
      Estimate Std. Error   z value
-1|0  2.0580   0.3502      5.876
0|1   3.0389   0.3514      8.648
1|2   7.1628   0.3537      20.251
```

The Spearman Rank Correlations were computed as described in section 4.3.5.2 to determine if there were any strong correlations between the predictor variables. These statistics do not have associated p-values because most of the input variables are categorical.

Table C.6: Spearman's Rho statistic for correlations between all variables in the best model for open canopy woodland vegetation change. Variables are: Solar radiation (solar), time since fire (TSF), elevation (elevation).

Variable 1	Variable 2	Rho
TSF	Solar	-0.0504
TSF	Elevation	-0.0634
Solar	Elevation	0.2396

Chapter 5: Burn-P3

The following is additional information associated with Table 5.5 to describe the semivariogram models used to build the spatial interpolation for each contrast pair in the historical restoration scenario to predict vegetation in the non-visible portion of the landscape. The methods associated with this table are described in section 5.3.2.1.2.

Table C.7: Semivariogram model details used to create the spatial interpolations for predict the non-visible portion of the historical fuel grid. The first step in the Indicator Kriging model requires constructing a semivariogram of the known values (visible portion of the landscape) to determine the spatial relationship between the two categories in each contrast pair. details related to contrast pair interpolation probability surface generation. For_W = forest plus woodlands, NonForW = nonforest plus woodlands, CF = conifer plus woodlands, MG = meadows/grasslands, BD = broadleaf deciduous, MX = mixedwoods, WD = woodlands.

Test	Nugget	Type	Anisotropy	Major Range (m)	Minor range (m)	Direction (Azimuth)	Partial Sill	Lag size	# of lags	Samples
For_W Vs Nonfor	0.0677	Exponential	No	948.8	n/a	n/a	0.148	100	12	33338
For Vs NonForW	0.0572	Exponential	No	771.9	n/a	n/a	0.161	100	12	33338
CF	0.0417	Exponential	Yes	1200	734.3	177.9	0.111	100	12	33338
MG	0.0677	Exponential	No	948.8	n/a	n/a	0.148	100	12	33338
BD	0.0	Stable	Yes	703.6	486	5.6	0.118	100	12	33338
MX	0.02	Stable	Yes	1166.3	496.5	8.1	0.023	100	12	33338
WD	0.0339	Stable	No	655.3	n/a	n/a	0.046	100	12	33338