<u>The emergence of novel disturbance in Jasper National Park – evaluating the causes and implications of 100 years of landscape change using repeat photography</u>

by

James Tricker
B.A., Rhodes University, 2003
M.Sc., University of Leeds, 2009

A Dissertation Submitted in Partial Fulfillment
of the Requirements for the Degree of

DOCTOR OF PHILOSOPHY

in the School of Environmental Studies

We acknowledge and respect the Ləkʷəŋən (Songhees and Xʷsepsəm/
Esquimalt) Peoples on whose territory the university stands, and the
Ləkʷəŋən and W̱SÁNEĆ Peoples whose historical relationships with the
land continue to this day

The emergence of novel disturbance in Jasper National Park – evaluating the causes and implications of 100 years of landscape change using repeat photography

by

James Tricker
B.A., Rhodes University, 2003
M.Sc., University of Leeds, 2009

Supervisory Committee

Dr. Eric Higgs, Supervisor
School of Environmental Studies

Dr. Trevor Lantz, Departmental Member
School of Environmental Studies

Dr. Jeannine Rhemtulla, Outside Member
Faculty of Forestry, University of British Columbia

Abstract

Recurring disturbance has a strong influence on the bounds of ecosystem variability. The concept *historical range of variability* (HRV) describes these bounds, providing a sense of the range of ecosystem characteristics exhibited in response to disturbance and recovery over time and space. Altered and novel disturbances can drive changes in ecosystem composition and configuration that depart from the HRV and lead to regimes shifts. In Jasper National Park, a systematic set of historical and repeated oblique photographs depict montane landcover in the aftermath of extensive fires in 1915 and a mountain pine beetle (MPB) outbreak in 2020/22. However, the MPB disturbance is historically unprecedented, and raises important questions about whether the characteristics of this event are within the HRV of the montane ecosystems. The focus of this dissertation is to apply a new workflow for deriving landcover maps from oblique photographs to evaluate the landcover changes that have occurred in the park's montane ecoregion over the last 105 years. The workflow comprises a deep learning algorithm that automates the classification of landcover evident in grayscale and color oblique photographs and a georeferencing tool that incorporates these data into a GIS. I report on the accuracy of the data produced by the workflow (Chapter 2) and quantify the changes in composition and configuration of broad landcover types after the two disturbance events for a study area in the montane ecoregion (Chapter 3). A scenario planning exercise is then undertaken to evaluate the uncertainty surrounding the implications of these changes and the potential for future novel disturbance events (Chapter 4). Georeferencing accuracy using root-mean-square error for a subset of 7 images was 4.6 m and overall classification accuracy for the landcover map produced from oblique photographs using the new workflow was 68%. The change analysis in the montane ecoregion indicated that the MPB outbreak has returned a version of heterogeneity evident in 1915 to the landscape by reducing the dominance of mature conifer (both in composition and configuration) across the landscape. Four scenarios then describe alternative futures in the park based on different levels and combinations of ecological novelty and management intervention. The value of this research is to validate the development of a new workflow for analyzing historical and repeat photographs, increase the temporal depth of ecological monitoring in the park, and allow managers and restoration practitioners to develop a better understanding of how and where novel disturbance is altering ecological processes and could reoccur in the future.

Table of Contents

List of tables

List of figures

Acknowledgements

First and foremost, I am deeply indebted to the guidance and support of my supervising professor, Dr. Eric Higgs. Over the course of my doctoral research, I have explored many wild places (both literally and figuratively), for which your trust, patience and encouragement has been ever-present. Thank you for taking a chance on me, Eric – this research project has been one of the most rewarding experiences of my life.

My supervisory committee, Drs. Jeanine Rhemtulla and Trevor Lantz, have been instrumental in the development of the research undertaken in this dissertation. Thank you both for your valuable insights and steady encouragement.

Under the broad MLP umbrella exists an eclectic mix of highly knowledgeable and experienced characters that have influenced my research project in innumerable ways. Julie Fortin's preceding graduate research was inspirational and her knowledge (and documentation) of all things related to repeat photography was invaluable as I entered the MLP fold. I'm grateful to Mary Sanseverino's infectious enthusiasm for mountain landscapes (not that I needed any encouragement!), as well as her technological bent regarding camera operation and photo interpretation. I'm indebted to Mike Whitney for his innovative advancement of monoplotting, which became a key part of my research methods. Similarly, it was rewarding to have a small hand in Spencer Rose's impressive graduate work on automated image classification, which yielded another key tool for my research methods. The process of developing these tools into a viable workflow and analyzing the output has greatly benefited from the technical insights of Claire Wright, who was always available to jump on a call at short notice. Rob Watt's encyclopedic knowledge of both the MLP image collection and the Canadian cordillera is infinite, and I remain in awe of his knack for precisely locating camera stations. Rick Arthur's considerable experience had a soothing influence on fieldwork operations, and his homemade pizzas were always a highlight of the field season. I'm grateful to the ever-dependable Jill Delaney and the teams she worked with at Library and Archives Canada for delivering digital copies of the remarkable survey images that I have been so fortunate to work with. And finally, thank you to Kate Fryer and Alina Fisher for all your hard work behind the scenes.

It has been a privilege to focus my research project in Jasper National Park. Memories abound from exploring and working in this extraordinary landscape. I'm also thankful to the many Parks Canada personnel who were instrumental in providing research permits, in-kind support, and fieldwork assistance, specifically, Rachel Dietrich, Chris Watson, Landon Shepherd, and the one and only Mike Eder.

I have been fortunate to receive generous financial support during my PhD. Thank you to Keith McClain at fRI, Nolan Beise at Mitacs and the University of Victoria.

I'm grateful for the support and camaraderie of my peers in the Visualization Lab and graduate cohort. Within this group, fieldwork has provided a unique opportunity to really get to know some of you. These shared experiences, often involving long days of hardship, make friendships for life. Thank you Sonia Voicescu, Cass Bunnk, Charles Hayes, Kris Lang, Claire Wright, Astra Lincoln, Suzie Zak, Shima Tajarlou, Cata Madrid, Alina Fisher, and Sarah Jacobs for the wonderful memories – I'm already looking forward to reminiscing will all of you.

And finally, to my friends and family – you know who you are. Thank you for being there for me during this crazy journey – I owe each of you many hugs and beers.

Dedication

To my parents, who have always believed in me.

And to the wild places, which always inspire me.

Chapter 1: Introduction

1.1 Background

Two significant ecological disturbance events have bookended my experience studying landcover change in Jasper National Park (hereafter "Jasper NP"). My first visit to the park in early July 2019 occurred at the tail end of a decade-long mountain pine beetle (MPB; *Dendroctonus ponderosae*) outbreak. The purpose of the visit was to undertake a repeat photography pilot study to document the changes the novel beetle epidemic was imparting on Jasper NP's montane ecoregion. I was accompanied by Drs. Jeanine Rhemtulla and Eric Higgs – two seasoned researchers with intimate knowledge of the park. Two decades earlier they had spent three arduous summers repeating hundreds of historical survey photographs from peaks and promontories high above Jasper NP's broad valleys. I joined in their astonishment as we viewed the considerable extent of red-tinged forest killed by the beetle from the various vantage points we visited to repeat photographs.

Five years later, I awoke to the devastating news that the town of Jasper had been caught in the path of the Jasper Wildfire Complex (JWC). The wildfire first sparked to life on July 22, 2024 from multiple lightning strikes north and south of the town. Driven by strong winds and dry fuels, the fires grew rapidly to 6,000 ha within hours, which overwhelmed initial attack crews and closed two key travel routes in the park. This fast-moving situation prompted evacuation orders for the 25,000 residents and visitors in the park via the one remaining route out of the park. Aggressive fire behaviour continued the next day as the fire doubled in size and continued to thwart suppression responses. Within forty-eight hours the wildfire had doubled in size again and reached the west end of town. Structural losses were catastrophic, with up to a third of buildings razed or damaged within the townsite. Outlying commercial accommodations, Parks Canada facilities and visitor amenities also sustained significant damage. It took until September 7th before Parks Canada could finally classify the JWC as under control, by which time it had burned 32,722 ha.

The occurrence of these two disturbance events can in part be explained by a lack of disturbance over the previous century. Decades of fire suppression altered the historical fire regime (characterized by frequent, mixed-severity fire) in Jasper NP (Chavardes and Daniels 2016; Tande 1979). In the absence of fire, closed-canopy forests increased in extent, with a majority of stands dominated by coniferous species (Rhemtulla et al. 2002). The abundance of mature lodgepole pine

(*Pinus contorta*), coupled with favourable climatic conditions, were key drivers of the historically unprecedented MPB outbreak (there are no previous records of MPB in Jasper NP prior to 2010; Dalman 2004). Similar factors contributed to the high intensity and severity of the JWC. The potential for fundamental ecological change (i.e., regime shifts) in response to these disturbances is now a pressing concern for Jasper NP.

The aim of this dissertation is to evaluate the impacts of altered disturbance regimes and novel disturbance agents on the historical range of variability (HRV) of montane landcover in Jasper NP. HRV relates to the bounds of system behavior that emerge over time as ecosystems adapt to recurring disturbance and has applications for natural resource management (see Section 1.4.2). Assessing changes to HRV requires temporally deep and spatially explicit data. Historical survey photographs captured in 1915 provide insights into the landcover present in Jasper NP under the historical fire regime and can be used as a unique baseline for quantifying changes to HRV. A new approach for producing landcover maps from oblique photographs is developed and tested in Chapter 2. This approach is then deployed in Chapter 3 to quantify the changes between the historical and contemporary photographs and assess how novel disturbances are altering the HRV of montane ecosystems in Jasper NP. Finally, to address the uncertainty concerning ecological responses to continued disturbance novelty, a plausible set of future scenarios are developed in Chapter 4 to critically assess the evolving value of HRV and support decision-making for scientists and managers who are contending with increasing novelty in wildlands and protected areas. The purpose for the rest of this chapter is to introduce my research questions and provide critical context for the work undertaken in this dissertation.

1.2 Description of research questions and projects

1.2.1 Chapter 2 - Assessing the accuracy of georeferenced landcover data derived from oblique imagery using machine learning.

Repeat photography offers distinctive insights into ecological change, with ground-based oblique photographs often predating early aerial images by decades. However, the oblique angle of the photographs presents challenges for extracting and analyzing ecological information using traditional remote sensing approaches. To address this need, this chapter introduces an end-to-end workflow that deploys two recently developed custom software tools: a trainable segmentation network and automated landcover classification algorithm, and a web-based georeferencing tool. To demonstrate the application of this workflow, a composite landcover map is produced for a study area in Jasper NP using images in the Mountain Legacy Project (MLP) collection. The purpose of this chapter is to assess the performance of the tools and evaluate the accuracy of the map in relation to landcover data produced using more conventional techniques (i.e., supervised classification of 30 cm aerial imagery captured over the study area in 2020).

1.2.2 Chapter 3 - A tale of two disturbances: Can mountain pine beetle restore landcover composition and pattern altered by fire suppression in Jasper National Park?

Altered and novel disturbances can drive changes in ecosystem composition and configuration, resulting in departures from their HRV. In Jasper NP, the historical and repeated survey photographs depict montane landcover in the aftermath of two extensive disturbance events. However, the recent MPB outbreak in the park is historically unprecedented and raises important questions about whether the characteristics of this event are within the HRV of the montane ecosystems. The focus of this chapter is to compare the composition and pattern of the montane ecosystems after the two disturbance events to better understand what impact the novel insect disturbance imparted on landcover in Jasper NP's montane ecoregion. To undertake this comparison, I employ the new workflow to produce historical (1915) and contemporary (2020-22) landcover maps for a study area in Jasper NP. The two maps are then used to 1) assess the landcover changes that have occurred in the park since fire suppression began and 2) consider whether novel insect disturbance can reintroduce landcover composition and pattern altered by fire suppression in Jasper NP's montane ecoregion.

1.2.3 Chapter 4 - Beyond history: Anticipating the influence of a novel disturbance regime on future landcover change in Jasper National Park.

The tragic 2024 Jasper Wildfire Complex is a recent example of a novel disturbance regime in Jasper NP. Anticipating ecological responses to novel disturbance regimes (and their interactions) is difficult and traditional management approaches (i.e., HRV) to maintain or restore ecological integrity may prove ineffective. In this chapter, I undertake a scenario planning exercise to conceive plausible futures in Jasper National Park. I developed four scenarios that emerge from the recent novel disturbance events based on different levels and combinations of ecological novelty and management intervention. The purpose of this planning exercise is to better understand how these two variables could influence the future of the montane ecoregion in Jasper NP and assess the continuing relevance of HRV in rapidly changing ecosystems.

1.3 Jasper National Park

Jasper NP comprises 11,000 km^2 of the Canadian Rockies in west-central Alberta, Canada. It is located within Treaty 6 and 8 lands, as well as the traditional territory of the Anishinabe, Aseniwuche Winewak, Cree, Nêhiyawak, Stoney Nakoda, Secwépemc, Dene-zaa, Mountain Métis and Métis Peoples. The park was established in 1907, and is the largest of a contiguous network of national and provincial protected areas located along the boundary between British Columbia and Alberta. Jasper NP is renowned for its exceptional landscapes, comprising soaring rugged peaks, extensive icefields and glaciers, alpine meadows, myriad lakes, rivers and waterfalls, and expansive biodiverse valleys. It is also a haven for the charismatic fauna found in the Canadian Rockies, including wolves, grizzly and black bears, wolverines, elk, moose, sheep, goats, and mountain caribou. The town of Jasper, originally developed as a railway community, is located at the intersection of the three main valleys in the park and has a local population of just under 5,000 residents. With upwards of 2.5 million visitors drawn to Jasper NP annually, the local economy is primarily focused on tourism.

The geographic location of Jasper NP and the steep altitudinal gradient of the mountainous terrain has a strong influence on the local climate and vegetation, resulting in three distinct ecoregions (Holland & Coen 1983). The lowest elevations comprise the montane ecoregion, where the climate is typically drier and warmer (Holland & Coen 1983). Lodgepole pine is the dominant tree species, although Douglas Fir (*Pseudotsuga menziesii*), white spruce (*Picea glauca*) and trembling aspen (*Populus tremuloides*) are also present (Holland & Coen 1983). Grasslands are interspersed among woodlands in the montane ecoregion, and wetlands are often found adjacent to water bodies (Beschta & Ripple 2007). The subalpine and alpine ecoregions occupy the higher elevations in the park, where the climate is colder and wetter (Holland & Coen 1983). The mid-elevation subalpine ecoregion typically comprises subalpine fir (*Abies lasiocarpa*), Engelmann spruce (*Picea engelmanii*) and lodgepole pine forests, transitioning into open shrub-forb meadows found among the rock and ice in the high-elevation alpine ecoregion (Bradley & Neufeld 2012).

1.4 Landcover change in Jasper NP over the previous century

Since time immemorial, a diversity of Indigenous Peoples, including the Anishinabe, Aseniwuche Winewak, Dene-zaa, Nêhiyawak, Secwépemc, Stoney Nakoda, Mountain Métis and Métis, inhabited the territory that encompasses Jasper NP. They developed unique and intricate ties to the territory, which was bound by respect and stewardship for the landscape (Parks Canada 2024). Indigenous land use fluctuated for centuries due to climatic variations and the cyclical patterns of game populations (Parks Canada 2024). However, the arrival of European explorers and trappers in the early 1800s and the establishment of a trading route through the region marked the beginning of change. By the end of the century, Indigenous and Métis families homesteaded in the Athabasca River valley, north of the present site of the town of Jasper. Following the establishment of Jasper NP in 1907, these families were forcibly removed, and the natural landscape was promoted for European settler recreation (MacLaren et al. 2005; Parks Canada 2024). In 1911, the Grand Trunk Pacific Railway reached the Jasper townsite, ushering in a new era of tourism in the park. Effective fire control in the valley began around this time to protect the transportation corridor and recreation opportunities (Tande 1980).

The persistence of fire suppression policies in Jasper NP would continue for much of the 20th century. In the absence of fire, landcover in the Park's montane ecoregion has changed considerably. One of the first studies to quantify these changes was undertaken by Jeanine Rhemtulla using repeat photography. In the course of research associated with the University of Alberta "Culture, Ecology and Restoration" program and seeking ideas for a master's graduate thesis in 1996, Rhemtulla was shown a large collection of bound black-and-white photographs by a Park staff member. The images were captured from elevated positions in the park and showed detailed historical landcover, which appeared very different to the contemporary landcover of the Park. She soon learned that the images were captured in 1915 by a Dominion Land Survey (DLS) team led by Morrison Parson Bridgland for the purpose of producing the first topographic maps for the central portion of the Park.

1.4.1 Dominion Land Survey, phototopography and repeat photography

In the late 1800's, the DLS had developed and began employing a new method called phototopography for mapping Canada's vast mountainous regions (MacLaren et al. 2005). The technique involved capturing a systematic set of panoramic images with a specially designed camera from mountain peaks and high points. At each camera station, the altitude and geometric measurements (to other camera stations nearby) were also recorded. Once the fieldwork was completed, the images, captured on 4x6 inch glass plates, and measurements could then be used to create topographic maps. For producing maps for the 1915 Jasper National Park survey, Bridgland's team established ninety-two camera stations and captured a total of 735 images. The unintended legacy of these photographs is their enduring record of historical landcover in the montane ecoregion prior to fire suppression and less than a decade after the Park's establishment.

Land-based repeat photography is a method for assessing long-term ecological change (Webb et al. 2010). Essentially, it involves taking photographs of the same scene from the same location at different points in time (Hastings & Turner, 1965; Rogers et al., 1984). One significant advantage of this method is that historical photographs often predate nadir imagery (e.g., aerial photographs or satellite images) by decades and capture western North American landscapes before the widespread effects of colonial and settler change (Morgan et al., 2010). This temporal depth provides a distinctive baseline for evaluating ecological changes over extended periods. However, repeat photography has limitations. The oblique angles of land-based images render them incompatible with classification and analysis techniques developed for nadir imagery. As a result, early repeat photography studies were initially limited to subjective descriptions of change.

After repeating all 735 images from Bridgland's Jasper National Park survey, Rhemtulla undertook a unique change analysis for a study area in the lower Athabasca River valley (Fig. 1.1). Using 20 pairs of historical and repeat photographs, Rhemtulla et al. (2002) manually delineated and classified areas of homogenous cover in a manner like aerial photographic interpretation (e.g. using transparent acetate overlays on hard copy photographs). The classified repeat photographs were successfully compared against land cover classes derived from contemporary aerial photographs (taken in 1991) to validate the approach (Rhemtulla et al. 2002). Subsequent analysis of the classified photograph pairs produced relative change matrices that indicated vegetation changes characterized by a shift toward late-successional vegetation types dominated by closed-canopy coniferous stands (Rhemtulla et al. 2002).

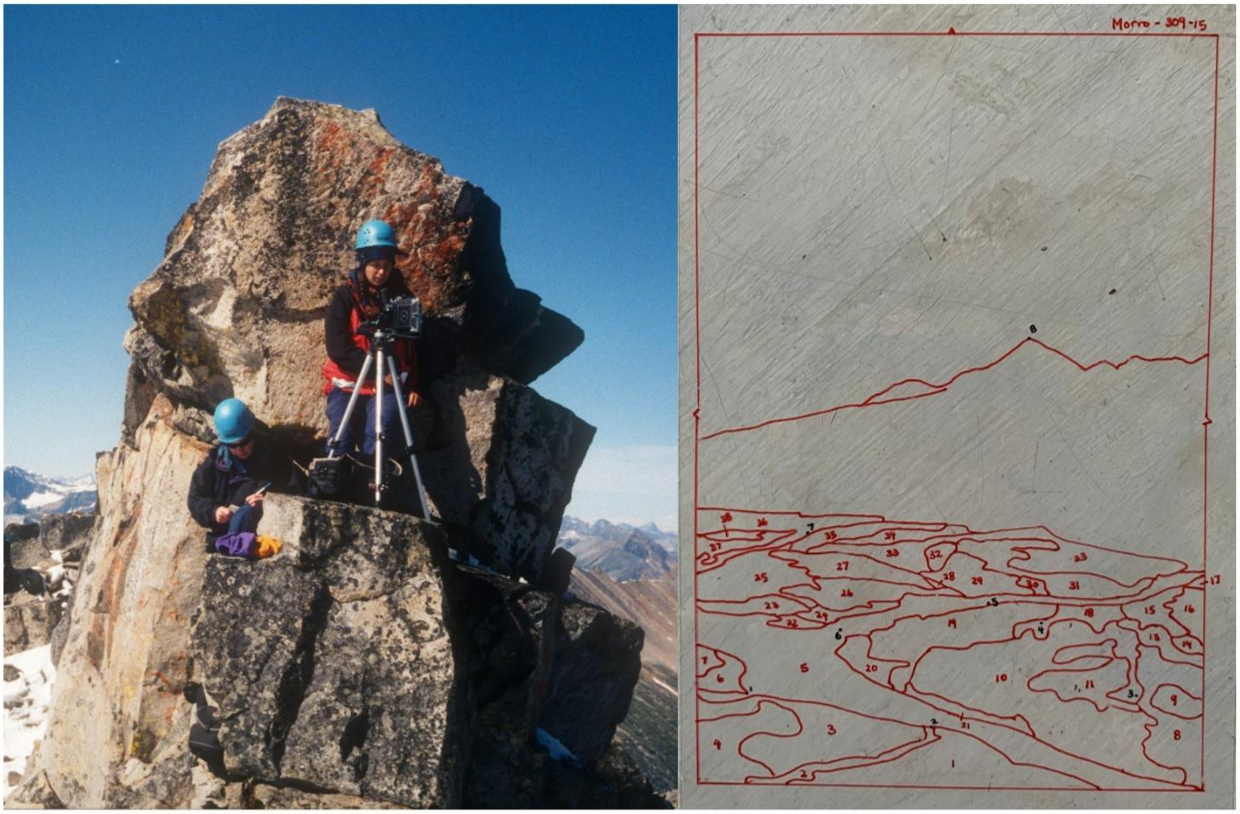


Figure 1.1 Left: Jeanine Rhemtulla (right) and Eric Higgs repeating historical photographs atop Mt Henry, Jasper National Park in 1999. Right: Acetate overlays of landcover data produced by Rhemtulla et al. 2001.

1.4.2 Historical range of variability

Ecosystems in Jasper NP's montane ecoregion vary over space and time in composition, structure, and processes. The concept of historical range of variability (HRV) was developed to describe the history of ecosystems and "define the bounds of system behavior that remain relatively consistent over time" (Morgan et al. 1994, pg. 88). Thus, the HRV concept allows for an improved understanding of changes occurring in ecosystems and provides managers with a framework for setting goals that are more likely to maintain and protect them (Landres et al. 1999, Keane et al. 2009). Specifically, the rate of change in ecosystem characteristics is an important consideration for managers, as it affects the ability of species to adapt to new conditions (Morgan et al. 1994).

Studying HRV improves ecosystem management by providing an understanding of ecosystem processes and species adaptions to change and can be used as a reference for establishing the range of desired future conditions (Morgan et al. 1994). However, criticism of HRV has focused on the relevance of the concept to resource management and ecological restoration in a world where present and impending ecological changes are unprecedented (Harris et al. 2006; Millar et al. 2007; Stephenson et al. 2010). This criticism would be valid if the only application of HRV was to restore ecosystems to replicas of pre-European conditions. Instead, Romme et al. (2012) argue that the evolving value of HRV is to use the historical record to assess how ecosystems "responded to environmental variation in the past as a way of informing current and future conservation management" (pg. 11). For example, managers can use an understanding of ecosystem dynamism provided by HRV to move an ecosystem back within the bounds of various discrete parameters (Higgs et al. 2014).

The quantification of HRV requires temporally deep (i.e. > 100 years) and spatially explicit historical data (Keane et al. 2009). Historical vegetation conditions can be reconstructed or described from pollen deposits found in lake or ocean sediments; plant macrofossil assemblages preserved in middens, sediments, or soil; dendrochronological data and fire scar histories, land survey records and repeat photographs (Humphries and Bourgeron 2001; Keane et al. 2009). However, these datasets present many challenges for quantifying HRV as they have limited or unknown spatial domains or their time spans are too long for assessing patterns of structure and composition (Swetnam et al. 1999). Keane (2013) asserts that the best sources for quantifying HRV are spatial chronosequences or digital maps of historical landcover from different time periods. Historical and repeated photographs have potential to meet these requirements.

1.4.3 Advances in repeat photography analysis

Numerous methods and tools for deriving landcover from oblique images have been developed since the pioneering work of Rhemtulla et al. (2002) in Jasper NP. The following studies have sought to improve the four essential repeat photography tasks for oblique images: image alignment, classification, georectification, and mosaicking. Roush et al. (2007) quantified vegetation changes within the alpine tree-line ecotone of Glacier National Park by orthorectifying image pairs in a GIS and overlaying a grid on each image to depict differences on a cell-per-cell basis. Gat et al. (2011) investigated the use of feature detection software to automate the process of aligning image pairs and Jean et al. (2015) tested an algorithm utilizing two different texture descriptors to classify image pairs into forest and non-forest categories. Stockdale et al. (2015) described a new method for georeferencing oblique photographs in a GIS using the WSL Mono-plotting tool (WSL is the acronym for the Swiss Federal Institute for Forest, Snow and Landscape Research) developed by Bozzini et al. (2012).

In 2016, Sanseverino et al. introduced the Image Analysis Toolkit (IAT), which provides a variety of tools for aligning and manually classifying image pairs, produces comparative statistics for user-defined categories, and allows different ways to visualize change. Fortin et al. (2019) compared the quantification of land cover from oblique photographs and orthogonal satellite imagery using regression analysis, finding a positive relationship between the categories in the two data sources. Most recently, Bayr and Puschmann (2019) demonstrated the application of a convolutional neural network for the automatic detection and classification of woody regrowth vegetation in repeat landscape photographs. However, despite the impressive progress reported in these studies, there remains a need for a workflow that seamlessly incorporates all four essential tasks for repeat photography analysis to support studies that quantify HRV.

1.4.4 The Mountain Legacy Project and software development

Many of the repeat photography studies described above made use of images from the MLP (https://mountainlegacy.ca/), directed by Dr. Eric Higgs, which provides access to the world's largest collection of systematic historical and repeated mountain photographs. Based in the School of Environmental Studies at the University of Victoria, the MLP originated from the work of Rhemtulla et al. (2002) and is focused on studying landscape change in the Canadian cordillera. This work includes repeat photography, archival research, data management, image analysis, and software development. This latter aspect of the MLP has benefitted from three important contributions in recent years that support the realization of an efficient and accurate repeat photography analysis workflow.

The first contribution, by Spencer Rose, a graduate student in UVic's School of Computer Science, involves the development of a deep learning algorithm that addresses a critical need for repeat photography analyses – automating the classification of landcover in grayscale and color oblique images. Many repeat photography analyses (i.e., Rhemtulla et al. 2001; Fortin et al. 2019; McCaffrey & Hopkinson 2017; Stockdale et al. 2015) have involved time-consuming manual image interpretation. Previous work to automate oblique image classification produced coarse results or was limited to color images (Bayr & Puschmann 2019; Jean et al. 2015). Rose's graduate research focused on developing a trainable segmentation network and land cover classification tool - the Python Landscape Classifier (PyLC; Rose 2020). The tool utilizes a deep convolution neural network optimized for semantic segmentation and is trained on grayscale and colour photographs from the MLP collection and their corresponding land cover masks manually created by Fortin et al. (2019) and Jean et al. (2015).

The second contribution, by Mike Whitney and Mary Sanseverino, research associates with the MLP, involves the development of a new georeferencing tool for oblique images. This tool addresses a shortcoming of the WSL Mono-plotting tool developed by Bozzini et al. (2012), which can only georeference vector data. The new georeferencing tool utilizes a web-based ray tracing algorithm, camera metadata and elevation data to relate landcover classifications derived from an oblique photograph to its correct geographic location in a GIS (Higgs et al., 2020). Importantly, this approach enables the georectification of rasterized classification data, which better supports the production of historical and contemporary landcover maps for the change analysis undertaken in this dissertation.

More recently, Wright et al. (2024) introduced the Mountain Image Analysis Suite (MIAS) plugin for QGIS (an open-source GIS application). The plugin integrates PyLC and the georeferencing tool into a single platform with a user-friendly interface. Consequently, MIAS significantly improves the efficiency and useability of the workflow developed for the analysis undertaken in this dissertation.

1.5 Drivers of change in Jasper NP

The two recent disturbance events described at the beginning of this chapter have abruptly reversed the gradual landcover changes observed by Rhemtulla et al. (2002) in the montane ecoregion over the previous century. The MPB outbreak has impacted up to 229,000 hectares (approximately 70 % of the total majority lodgepole pine stands within the park) over the previous decade (Skretting et al. 2024). In 2024, the high severity JWC burned through 32,722 ha of the upper Athabasca River valley. Approximately 70% of the area burned by the fire was in stands impacted by MPB. Although both disturbance events are naturally occurring (MPB is native to forests in other areas of the southern Canadian Rockies), anthropogenic drivers have likely influenced their extent and severity (i.e., MPB is unprecedented in Jasper NP). A driver refers to any natural or human-induced change in an ecosystem (Carpenter et al. 2006). Direct drivers are physical or biological and influence ecosystem processes, whereas indirect drivers may alter one or more of the direct drivers (Nelson et al. 2006).

A 310-year fire history chronology established by Tande (1979) for a 43,200 ha area centered on the town of Jasper determined that the montane ecoregion experienced a mixed-severity fire regime. From 1665 to 1913 there were 46 fires within the study area with a mean fire return interval of 5.5 years (Tande 1979). Since 1913 (through 1975) fire periodicity and extent declined significantly (Tande 1979). Another fire history chronology by Chavardes and Daniels (2016) focused on a 2,000 ha area 12km north of the town of Jasper. Across a similar period, they also found evidence of frequent, mixed-severity fire for numerous sites within their study area, but no fires after 1905 (Chavardes and Daniels 2016). Both studies suggest the decline in fire over the previous century is a result of fire exclusion and suppression.

Fire exclusion began in Jasper NP with the displacement of Indigenous and Métis families shortly after the establishment of the park in 1907. For millennia, Indigenous Peoples used fire as a tool for resource management (i.e., maintenance of travel corridors, improving forage for domestic animals, increasing availability of culturally important plants) and community protection (Hoffman et al. 2022). Consequently, Indigenous fire stewardship enhanced ecosystem diversity and reduced wildfire risk by lessening fuel loads and diversifying the frequency, timing and severity of wildfire (Christianson 2014). In 1913, Jasper NP implemented a policy of fire suppression to protect infrastructure in the Park and preserve European wilderness values (MacLaren et al. 2007; Murphy 2007). In the ensuing decades fire suppression responses greatly improved across the Canadian Rockies due to advances in fire-fighting equipment, the construction of fire roads, and the installation of fire lookouts (Keane et al. 2002; Woodley 1995). Between 1935 and1995, the annual area burned in the Canadian Rockies was approximately three percent of the previous, long-term average (Woodley 1995).

Another direct driver influencing disturbance is climate change. Many disturbances have a strong climate forcing, including wildfire and insect outbreaks (Turner 2010; Turner and Seidl 2023). In western North America, climates have become significantly warmer and drier over the last 50 years (Whitman et al. 2022). The implications for wildfire are longer fire weather seasons, more frequent extreme fire weather events, faster fire spread rates, and an increase in lightning-induced ignitions

(Coop et al., 2020; Jones et al., 2022; Pérez-Invernón et al., 2023). As a result, many regions are now experiencing an increase in large fires and total area burned, as well as higher fire severity (Abatzoglou & Williams, 2016; Bedia et al., 2015; Buch et al., 2023; Joseph et al., 2019; Krawchuk et al., 2009; Parks & Abatzoglou, 2020; Wasserman & Mueller, 2023; Whitman et al., 2022).

Some forest insects respond rapidly to climatic variation due to their sensitivity to temperature, which can affect their distribution and abundance (Weed et al. 2013). Historically, sustained cold winter temperatures in the northern Canadian Rockies have served as a geoclimatic barrier that has confined MPB to the western half of the continent (de lay Giroday et al. 2012). However, the warming trend across western North America has improved overwintering brood survival, flight capacity and generation times (Pureswaran et al. 2018). Carroll et al. (2004) attribute the recent range expansion of MPB into formerly climatically unsuitable habitats (i.e., northward, eastward, and higher elevations) to the effects of climate change (also see Bentz at al. 2016). Climate change has also been shown to double the length of the MPB flight season and increase the life cycle in some MPB broods from one to two generations per year (Mitton & Ferrenberg 2012). Additionally, changes in climate can lead to drought conditions that reduce host tree defences (Fettig et al. 2013; Seidl et al. 2017).

The synergistic effects of these drivers are causing disturbance regimes to depart from their HRV or occur in areas where they have not been present historically (Turner & Seidl 2023). In Jasper NP, the build-up of hazardous fuels over the previous century due to fire exclusion and suppression has resulted in higher severity fires. This departure from a mixed-severity fire regime has been amplified by the effects of climate change, such as the increased frequency of extreme fire weather events, which has made firefighting efforts exceedingly difficult.

Altered fire regimes and climate change across western Canada have also created ideal conditions for MPB population growth. A reduction in wildfire has led to a threefold increase in host tree availability across the region and warmer summers and milder winters have increased beetle reproduction and overwintering brood survival respectively (Taylor & Carroll 2003; Carroll et al. 2006). In combination, these factors were the catalyst for the largest recorded MPB outbreak in history, which began in central British Columbia in the late 1990's and enabled range expansion into northern BC and northern and central parts of western Alberta (including Jasper NP).

1.6 Novel disturbance regimes and their implications

The emergence of novel disturbance regimes, described by Turner & Seidl (2023) as disturbance regimes to which ecosystems are not adapted, are raising concerns about how ecosystems will respond. This is because disturbances and recovery processes are closely linked, so changes that alter this linkage could result in novel ecological responses (Turner 2010; Seidl & Turner 2022). Additionally, interactions between two novel disturbances (and other drivers of change) may increase the likelihood of novel responses (Turner & Seidl 2023). However, the potential for fundamental ecological change (i.e., a departure from HRV) is greatest when novelty occurs in both the disturbance and the ecological response to the disturbance (Seidl et al. 2016; Turner & Seidl 2023). When an ecosystem can no longer absorb change (i.e., an ecological threshold is surpassed), a regime shift to an alternative stable state can occur (Beisner et al. 2003; Folke et al. 2004; Harris et al. 2013; Scheffer et al. 2001).

Lewontin (1969) is credited with introducing the idea that ecosystems often exhibit multiple stable states (i.e. basins of attraction), depending on environmental conditions (Beisner et al. 2003; Groffman et al. 2006, Harris et al. 2013; Holling 1973). Ecosystems can shift between states in two ways. The first involves internal factors whereby the ecosystem shifts from one state to another by a "sufficiently large perturbation applied directly to the state *variables"* (e.g. population densities; Beisner et al. 2003, p. 2). The second way involves external factors whereby "a change in the *parameters* that determine the behavior of the state variables and how they interact with each other" leads to a shift in states (e.g. nutrient cycles; Beisner et al. 2003, pg. 2). These are known as the "community" and "ecosystem" perspectives respectively (Beisner et al. 2003).

Resilience is an important feature of ecosystems and refers to the resistance of a system to state shifts (Harris et al. 2013; Holling 1973). It relates to two topographic characteristics of a basin that act to retain the system: the steepness of the slope and the width of the basin (Beisener et al. 2003). Therefore, a system with steeper slopes and/or a wider basin will require a larger perturbation to escape to a different basin (i.e., alternative stable state; Beisner et al. 2003). Feedback mechanisms form the basic dynamics for regulating the state of an ecosystem and are an important part of nonlinear responses to environmental change (Maxwell et al. 2017). Negative feedbacks provide stability to the system, whereas positive feedbacks stimulate change and can lead to alternative states (Capon et al. 2015; Harrison 1979; Scheffer et al. 2001). Regime shifts may occur for several reasons but always involve a shift in feedback mechanisms that maintain a state in one form or another (Capon et al. 2015; Harris et al. 2013; Scheffer et al. 2001).

1.6.1 Novel ecosystems and ecological novelty.

Similarities exist between the occurrence of regime shifts and “novel ecosystems” – a concept introduced by Hobbs and colleagues to describe the emergence of new biotic assemblages with no natural analogs (Hobbs et al. 2006; Hobbs et al. 2009). Anthropogenic drivers are the catalyst for novel ecosystems, which occur when a system surpasses a critical threshold that falls outside its HRV (Hobbs et al. 2013). However, a key difference between regime shifts (to alternative stable states) and the emergence of novel ecosystems is the irreversibility of the latter – transitions to novel ecosystems are usually permanent. Hobbs et al. (2013) differentiate between novel and hybrid ecosystems, which contain novel elements, but have the potential to recover their historical qualities.

There is also similarity in the mechanisms behind regime shifts and emergent “ecological novelty” - an umbrella term proposed by Heger et al. (2019) that encompasses the concepts of novel organisms, novel communities, novel ecosystems, novel interactions and novel abiotic conditions. The term allows for novelty to be addressed from descriptive, non-normative perspectives (e.g., both categorical and continuous measures of novelty). The organism-centered perspective of this term considers if focal species lack eco-evolutionary experience to adapt to changing conditions (akin to a shift in variables) whereas the site-specific perspective considers if contemporary conditions differ from historic conditions (akin to a shift in parameters; Heger et al. 2019).

In Jasper NP, a key issue with emergent ecological novelty in the montane ecoregion concerns the capability of management interventions to support ecosystem resilience and adaptation. The first priority of park management per the Canada National Parks Act (2000) is to maintain or restore “ecological integrity”, defined as the condition of ecosystem components, communities, and processes determined to be characteristic of its natural environment (Keenleyside et al. 2012; Parks Canada 2008). Novel disturbance regimes are a threat to ecological integrity as they may cause ecosystems to depart from their HRV and shift to different species assemblages and/or ecological functions (i.e., alternative stable states or novel ecosystems). Thus, the research undertaken in this dissertation is ultimately focused on evaluating how novel disturbance could impact ecological integrity in Jasper NP.

Co-authorship statement

Chapter 2 is published in Remote Sensing in Ecology and Conservation. JT, JR, TL, and EH designed the study, SR and CR developed the research tools, JT undertook fieldwork, developed the workflow and analyzed data, JT led the writing of the manuscript with input from the entire author team.

Citation: Tricker, J., Wright, C., Rose, S., Rhemtulla, J., Lantz, T., & Higgs, E. (2024). Assessing the accuracy of georeferenced landcover data derived from oblique imagery using machine learning. *Remote Sensing in Ecology and Conservation*, *10*(3), 401–415. https://doi.org/10.1002/rse2.379

Chapter 3 is published in Regional Environmental Change. JT, JR, TL, and EH designed the study, JT undertook fieldwork and analyzed data, JT led the writing of the manuscript with input from the entire author team.

Citation: Tricker, J., Lantz, T. C., Rhemtulla, J. M., & Higgs, E. S. (2025). A tale of two disturbances: Can mountain pine beetle restore landcover composition and pattern altered by fire suppression in Jasper National Park? *Regional Environmental Change*, *25*(1), 21. https://doi.org/10.1007/s10113-024-02356-8

Chapter 4 is in preparation for submission to Journal of Environmental Management. JT, JR, TL, and EH designed the study, JT developed the scenarios, JT led the writing of the manuscript with input from the entire author team.

Citation: Tricker, J., Rhemtulla, J., Lantz, T., & Higgs, E. Beyond history: Anticipating the influence of a novel disturbance regime on future landcover change in Jasper National Park. In prep.

Chapter 2: Assessing the accuracy of georeferenced landcover data derived from oblique imagery using machine learning

Abstract

Repeat photography offers distinctive insights into ecological change, with ground-based oblique photographs often predating early aerial images by decades. However, the oblique angle of the photographs presents challenges for extracting and analyzing ecological information using traditional remote sensing approaches. Several innovative methods have been developed for analyzing repeat photographs, but none offer a comprehensive end-to-end workflow incorporating image classification and georeferencing to produce quantifiable landcover data. In this paper, we provide an overview of two new tools, an automated deep learning classifier and intuitive georeferencing tool, and describe how they are used to derive landcover data from 19 images associated with the Mountain Legacy Project, a research team that works with the world's largest collection of systematic high-resolution historic mountain photographs. We then combined these data to produce a contemporary landcover map for a study area in Jasper National Park, Canada. We assessed georeferencing accuracy by calculating the root-mean-square error and mean displacement for a subset of the images, which was 4.6 and 3.7 m, respectively. Overall classification accuracy of the landcover map produced from oblique images was 68%, which was comparable to landcover data produced from aerial imagery using a conventional classification method. The new workflow advances the use of repeat photographs for yielding quantitative landcover data. It has several advantages over existing methods including the ability to produce quick and consistent image classifications with little human input, and accurately georeference and combine these data to generate landcover maps for large areas.

Introduction

Historical ground-based (i.e., oblique) landscape photographs are often overlooked as a source of information for studying ecological change (Keane et al., 2009; Trant et al., 2015; Webb et al., 2010). These photographs vary in coverage and quality, from snapshots in personal albums to extensive and systematic collections of images used for land surveying. What many early landscape photographs have in common is they serve as a record of ecosystems in most cases before the effects of pervasive human impacts (see Steffen et al., 2015). Oblique landscape photographs can provide researchers with a spatially explicit reference from which to evaluate contemporary landscape features and change, including, for example, species composition and configuration (Gruell, 1983; Hayward et al., 2012).

Methods for analyzing ecological change using historical oblique photographs were pioneered in the field of repeat photography: the practice of capturing photographs of a scene from the same location at different points in time (Hastings & Turner, 1965; Rogers et al., 1984). Repeat photography was first documented in central Europe, initially as a technique for tracking glacial change in the Alps, but it has since become an established method for documenting different types of ecological and landscape change around the world (Webb et al., 2010). In recent decades, repeat photography studies have developed a variety of approaches to elicit information from oblique images and evaluate changes that have occurred between the time periods captured in each photograph. From early qualitative observations describing geomorphological and vegetative changes on the landscape (e.g., Byers, 2007; Gruell et al., 1982; Hastings & Turner, 1965), methods have evolved to include innovative quantitative approaches that generate measurable change metrics (e.g., Bayr & Puschmann, 2019; Rhemtulla et al., 2002; Roush et al., 2007).

Researchers undertaking quantitative analysis in repeat photography studies must perform two critical tasks: (1) identify and/or classify landcover information in the sequence of images; and (2) calculate the amount of detectable change. Oblique landscape photographs present several challenges to this process including perspective distortion, high interclass variation, image quality issues, and varying scales relative to pixel size (i.e., a pixel in the foreground of an image will capture less geographic area than a pixel in the background; Bayr & Puschmann, 2019; Clark & Hardegree, 2005; Kull, 2005; Sanseverino et al., 2016). Additionally, the oblique perspective of ground-based photographs does not allow for analysis using classification and change-detection techniques developed for nadir imagery (Bayr & Puschmann, 2019). Instead, studies have largely relied on manual photointerpretation techniques for measuring change.

Several different quantitative approaches have been used to analyze oblique images. For example, Rhemtulla et al. (2002) followed common aerial photographic interpretation techniques (i.e., hand-drawn polygons on acetate sheets overlaid on printed photographs) to delineate and categorize areas of homogenous landcover for image pairs. These data were then imported into a geographic information system (GIS), which allowed the authors to estimate relative vegetative change (as a percentage of photograph area) by summing pixels in image pairs by cover type (Rhemtulla et al., 2002). Fortin et al. (2019) modernized this approach by creating digital copies of image pairs and then manually classifying features of interest using a digitizing tablet and the Image Analysis Toolkit (IAT; custom software specifically developed for analyzing repeat photographs; Sanseverino et al., 2016). Hall (2001) and Roush et al. (2007) followed a different approach that overlaid a grid on top of image pairs and determined changes to vegetation cover per grid cell. More recently, Trant et al. (2020) used a digitizing tablet to delineate the treeline in 81 image pairs to assess high elevation ecosystem change. Other approaches have used a combination of image pairs and field observations to produce quantitative data (Hoffmann & Todd, 2010; Masubelele et al., 2015; McClaran et al., 2010;).

The different approaches for classifying landcover and calculating change between historical and repeat photographs emphasize two important shortcomings. The first is a reliance on manual photointerpretation to identify and categorize landcover information. Although this technique is an accurate approach (Wulder, 1998), it can be difficult to replicate and produce consistent results due to the subjectivity of different interpreters (Wulder et al., 2008). Manual photointerpretation is also a laborious, time-consuming, and expensive undertaking (Green, 2000). Recent studies have investigated the potential to use machine learning methods to classify information in oblique image pairs. Jean et al. (2015) developed an algorithm for classifying oblique photographs based on texture analysis using a machine learning segmentation algorithm that delineated "meta-categories" of forest and non-forest. Bayr and Puschmann (2019) used a deep learning algorithm to classify and evaluate woody vegetation change in pairs of repeat color photographs. However, both studies generated image classifications with coarse resolution limited to just two landcover categories and encountered challenges related to classifying historical grayscale photographs

The second shortcoming concerns the difficulty of georeferencing oblique images, and therefore allowing absolute rather than relative quantitative comparison. To address this, Bozzini et al. (2012) developed the WSL Monoplotting Tool for georeferencing oblique photographs using photogrammetry methods, whereby each photographic pixel is plotted to its real-world location. Stockdale et al. (2015) used this tool to georeference landcover data derived from repeat photographs (manually interpreted using grid overlays) in a GIS at 100 m resolution. Recently, Bayr (2021) tested the accuracy of the monoplotting tool using a fine-resolution digital elevation model (DEM), reporting a mean displacement of 1.52 m for georeferenced points relative to their location in aerial photographs. However, the tool requires users to identify control points from features recognizable in both oblique photographs and nadir imagery, which can be difficult and time-consuming. This is especially true if the oblique photographs precede the nadir imagery by decades, which is often the case for historical images. Further, no repeat photography study has yet to develop a workflow that harnesses the potential of automated classification approaches in combination with georeferencing procedures to produce repeatable, accurate, and fine scale landcover change data.

To address this need, we have created an end-to-end workflow that deploys two recently developed custom software tools: a trainable segmentation network and automated landcover classification algorithm, and a web-based georeferencing tool. These tools were developed by researchers associated with the Mountain Legacy Project (MLP; http://mountainlegacy.ca/), a research team that works with the world's largest collection of systematic high-resolution historic mountain photographs (Sanseverino et al., 2016). In this paper, we provide an overview of the two original tools and describe how they are implemented in a workflow for oblique photography analysis. To demonstrate the application of this workflow we produce a composite landcover map for a study area in Jasper National Park using images in the MLP collection. Our main purpose in this article is to assess the performance of the tools and evaluate the accuracy of the map in relation to landcover data produced using more conventional techniques (i.e., supervised classification of 30 cm aerial imagery captured over the study area in 2020). The intent of this paper is to validate the new workflow for the purpose of producing landcover data from historical oblique photographs. This would benefit ecology and conservation by increasing the temporal depth of spatially explicit studies focused on understanding change dynamics in ecosystems and landscapes.

Study area

To assess the capability of the workflow to produce accurate georeferenced landcover data, we required a study area that had comprehensive oblique and orthographic photographic coverage. Jasper National Park extends over 11,000 km^2 of the eastern Canadian Rockies in west-central Alberta (Fig 2.1). Established in 1907, it is the largest of a network of national, provincial, and wilderness parks that stretch along the continental divide on either side of the provincial border between Alberta and British Columbia. In 1915, the central portion of Jasper National Park was systematically photographed by Morrison Parsons Bridgland for the Dominion Land Survey (MacLaren et al., 2005). Bridgland established a network of 92 camera stations, typically on mountain peaks or prominent outcrops, from which he captured a total of 735 photographs (Rhemtulla et al., 2002). The photographs, along with horizontal and vertical measurements recorded with a theodolite were then used to produce the first topographic maps for the area (Higgs, 2003; MacLaren et al., 2005). To the northeast of the Jasper townsite, several camera stations provide unobstructed and overlapping views of the Athabasca River valley, resulting in almost 100% coverage of the landscape (Fig. 2.1). This area was the focus of a repeat photography study by Rhemtulla et al. (2002) that evaluated 80 years of landcover change in the valley. In this paper we reanalyze the 2002 study area using new repeat photographs captured in 2020 from Bridgland's original camera stations.

Materials and Methods

Our workflow for producing landcover data features two new tools for oblique image classification and georeferencing. The Python Landscape Classifier (PyLC; https://github.com/scrose/pylc) is a trainable segmentation network that automates the classification of landcover types in grayscale and color oblique photographs (Rose, 2020). PyLC is based on an implementation of Deeplabv3+, a top performing deep convolution neural network (DCNN) optimized for semantic segmentation (i.e., each pixel in an image is assigned a class or label; Chen et al., 2017). The network was trained on 95 pairs of historical and repeat photographs from the MLP collection and their corresponding landcover classifications created by MLP researchers using manual classification (Fortin et al., 2019; Jean et al., 2015). PyLC classifies images into 8 landcover classes originally defined by Jean et al. (2015) and based on the broad habitat categories found in the Canadian Rockies (Fig. 2.2; Table 2.1). Data augmentation (i.e., geometric manipulations of training samples) was used to increase underrepresented classes in the training dataset to mitigate class imbalance (Table 2.1).

This technique produced modest improvements to overall accuracy (+3% historic/+1% repeat), but much larger gains for underrepresented classes (Rose, 2020). Rose (2020) reported that the top performing models produced overall weighted F1 scores (a measure of the overlap between a segmentation mask and a manually classified mask) of 0.841 for historical photographs, and 0.909 for contemporary repeat photographs.

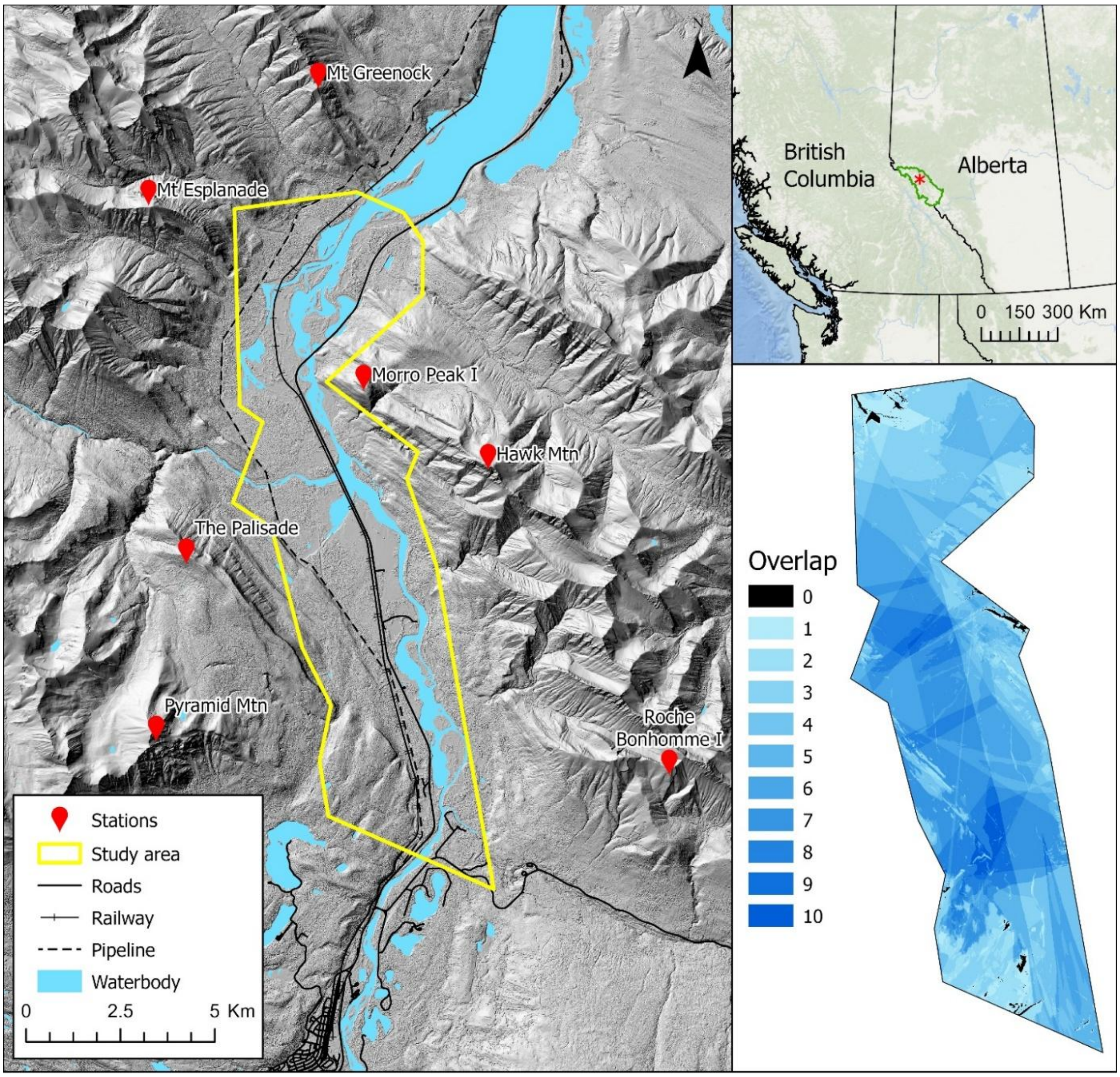


Figure 2.1 Map of the Athabasca River valley study area and camera station locations in Jasper National Park. Top right inset: location of study area (red star) and Jasper National Park (green) in Western Canada. Bottom right inset: Amount of field of view overlap for the 19 images used in the analysis. Map credits: ESRI, 2021; Natural Resources Canada, 2021.

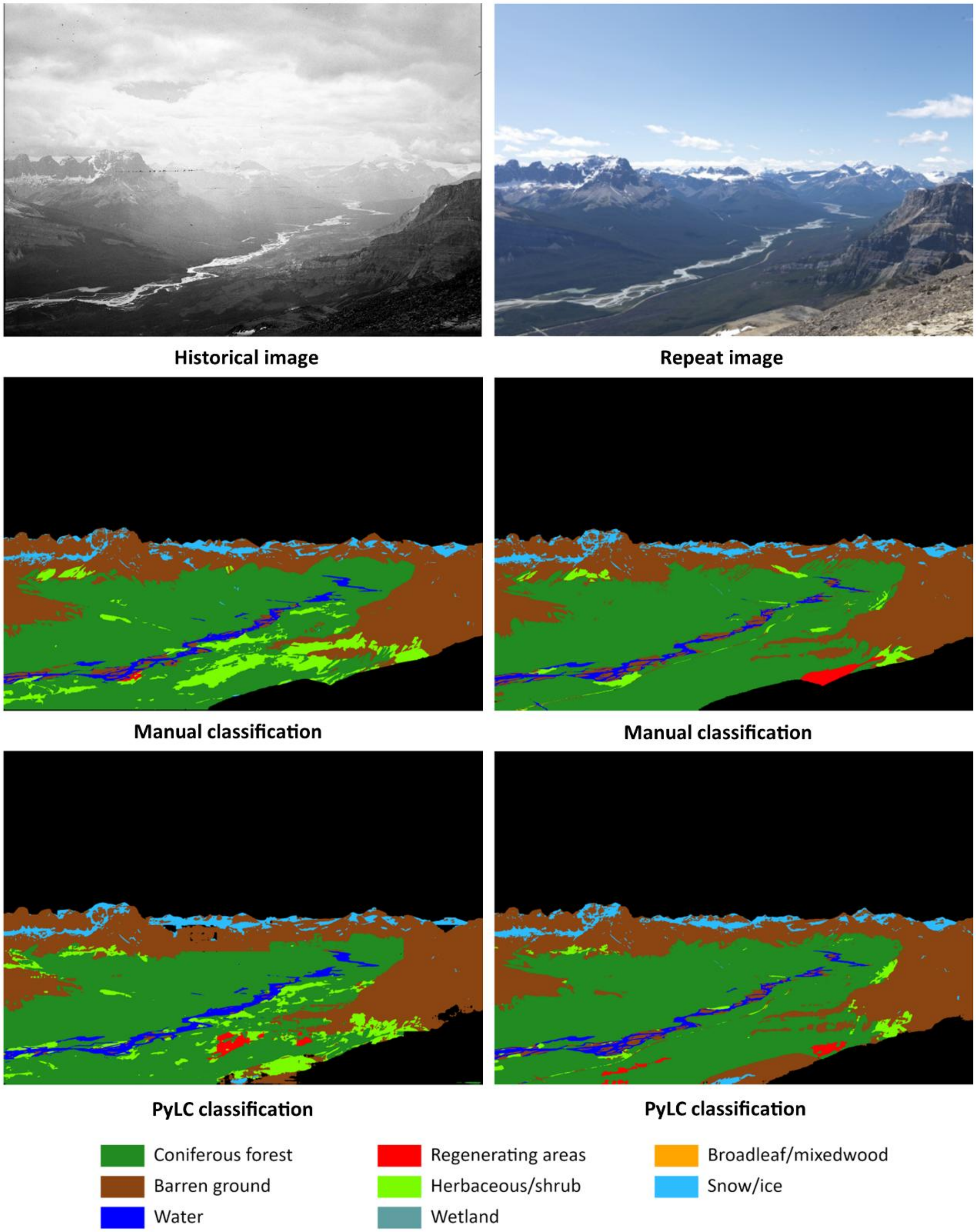


Figure 2.2 Examples of image classifications produced by manual methods and PyLC for historical and repeat photograph pairs. Classification accuracy was typically better for repeat photographs due to the higher image quality and richer spectral information. Figure adapted from Rose (2020).

Table 2.1 Description of land cover categories for PyLC/SVM classification scheme. Percent change indicates adjustments to pixel class distribution after data augmentation. SVM samples indicate the number of training samples per class used by the SVM classifier to generate the orthogonal landcover map. Adapted from Rose 2020.

Category	Description	Pixel distribution	Augmented pixel dist.	Percent change	SVM samples
Coniferous forest	Greater than 75% coniferous trees	0.219	0.2431	11	20
Barren ground	Soil, sand, gravel, or rock	0.1155	0.1149	−0.5	44
Water	Lakes and rivers	0.0008	0.0042	401	94
Regenerating area	Visibly recently burned forest	0.021	0.0748	255.5	N/A
Herbaceous/shrub	Shrubs, grasses, and herbaceous vegetation	0.0613	0.0638	4.1	49
Wetland	Vegetation with a wet or aquatic moisture regime	0.0082	0.0388	370.8	22
Broadleaf/ mixedwood	>70% broadleaf trees/patches with 30%–70% broadleaf cover and the rest coniferous trees and/or shrub	0.0095	0.0493	419.4	44
Snow/ice	Snow and ice	0.008	0.0203	155.9	N/A
Non categorized	Non categorized	0.5568	0.3909	−29.8	N/A

The georeferencing tool uses a new approach to relate landcover classifications derived from an oblique photograph to its correct geographic location (Higgs et al., 2020). The tool utilizes a web-based ray tracing algorithm to establish a correspondence between each image pixel and DEM cell. The tool requires information from the original photograph (camera location, azimuth of the camera orientation, the lens field of view (FOV), and the image dimensions) and a fine-resolution elevation data. The landcover classification from the photograph is georeferenced in three steps (Fig. 2.3). The algorithm generates a “virtual” version of an oblique photograph by tracing rays from the camera FOV to visible pixels (viewed as virtual columns) in a DEM. The original photograph is then aligned to the virtual photograph using control points and an perspective transformation. Finally, the algorithm exploits the relationship between the classified image (which shares the same dimensions as the original photograph) and the DEM to plot the landcover values in real-world space.

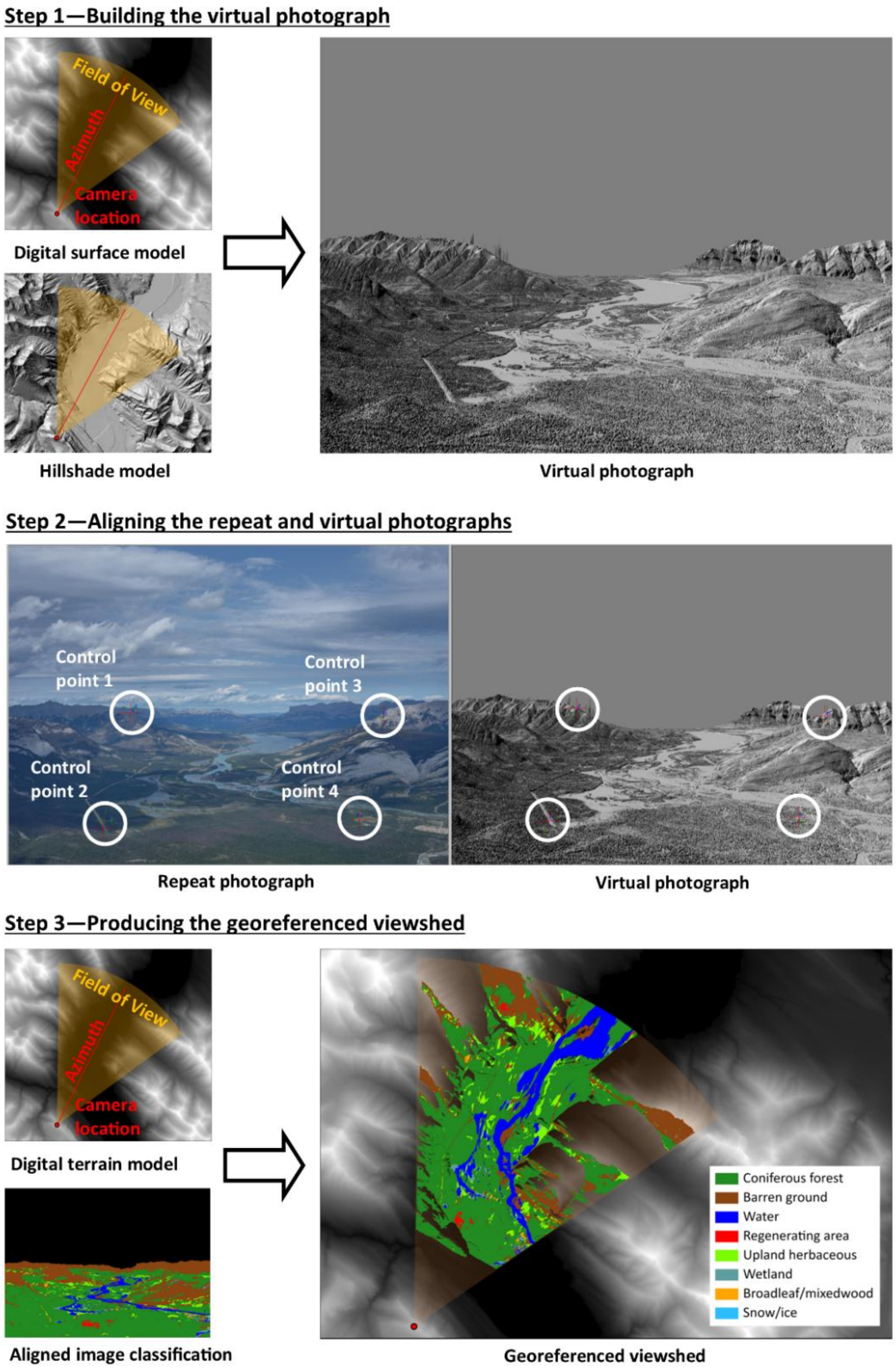


Figure 2.3 Steps for georeferencing classified photographs in IAT. Step 1: a virtual photograph, replicating the original photograph FOV, is produced using camera metadata and a DEM. Step 2: the original photograph is aligned to the virtual photograph using control points and a perspective transformation, which are then used to align the image classification to the virtual photograph. Step 3: the cell values from the aligned image classification are then georeferenced by exploiting the relationship between the virtual photograph and elevation data.

To assess the capability of PyLC and the IAT georeferencing tool to produce accurate landcover data, we undertook fieldwork in August/September 2020 to capture third sequential repeat photographs (original 1915; first repeats 1998/99) overlooking the study area. We required new repeat photographs because the first repeats were captured on black and white film and do not have the required spectral information for analysis using the current version of PyLC. Photographs were taken with a 51.4-megapixel FujiFilm GFX50s medium format digital camera with a 32–64 mm F4 zoom lens, and a Novoflex panoramic head and tripod. Field notes and location photographs (i.e., for camera tripod placement) recorded by Rhemtulla et al. (2002) were instrumental in planning and navigating to original station locations. Gridded printouts of the historic photographs were used to finetune the tripod position and align each repeat photograph through the camera viewfinder. Detailed field notes including coordinates of camera location using a Garmin GPS and azimuth for each photograph using a Brunton transit were recorded to assist with later georeferencing tasks. Seven survey stations, each comprising multiple photographs (5 to 12 photos per station), were revisited yielding a total of 69 repeat photographs (Fig. 2.1).

High-resolution elevation data derived from airborne LiDAR were obtained from Natural Resources Canada (2021) and Jasper National Park. These data were formatted, mosaicked, and resampled to 2m resolution to produce a 30 × 40 km digital surface model (DSM) and digital terrain model (DTM) for the study area (all GIS tasks were performed in ArcGIS Pro 2.8.3; ESRI, 2021). Note that the DSM produces more realistic virtual photographs by including surface features such as trees, which can be used for alignment with the same trees in the oblique photograph. The DTM is then used for georeferencing the classified images (avoiding the speckle effect that would be present if using the DSM). To determine which station photographs to include in this analysis, viewsheds were generated for each of the 69 photographs using the Viewshed tool (ESRI, 2021). Only image viewsheds that had a considerable spatial footprint within the study area were selected, which reduced the total number of images to 19 (providing 99.3% coverage of the study area; Fig. 2.1; Appendix A). We then classified landcover in each of the 19 images with PyLC using the landcover classes listed in Fig 2.2 (Appendix B).

The 19 classified images produced by PyLC were then georeferenced with the IAT georeferencing tool using the elevation data and camera metadata (see steps described in Fig. 2.3; Appendix C). To account for overlap between the georeferenced images, all 19 viewsheds were combined in ArcGIS

Pro 2.8.3 using the cell statistics tool (ESRI, 2021) with the majority overlay statistic selected (i.e., the value that occurs most often for cells with multiple values). Where cells had multiple majority values, the tool produced a no data value, leading to only 91.7% coverage of the study area. To address this, classified images were dropped iteratively based on a visual assessment of PyLC output quality with the majority cell value recalculated each time until two viewsheds remained. The resulting 19 grids were then mosaicked together in the order they were produced using the Mosaic to New Raster tool (ESRI, 2021). This process brought the coverage of the study area back up to 99.3%. The remaining unclassified cells were filled using their nearest neighbor values with the Nibble tool (ESRI, 2021). Several areas of water were misclassified as snow/ice in the study area – these areas were reclassified back to water using the Reclassify tool (based on the assumption that there is no ice in the valley bottom during the summer months; ESRI, 2021). Similarly, areas of regeneration were reclassified to conifer as there have been no recorded wildfires within the study area for decades. The final step for producing the oblique landcover map involved clipping the grid to the study area extent (Appendix D).

The accuracy of the IAT georeferencing tool was assessed using one georeferenced image from each of the seven camera stations. Eight test points were manually digitized on each georeferenced image using identifiable features within the study area. Control points for the same features were then digitized on 30 cm RGBI high-resolution aerial imagery obtained from Jasper National Park captured over the study area in 2020. We determined georeferencing error by calculating the root-mean-square error (RMSE) for each set of test and control points. To compare accuracy with the WSL tool, we also calculated the point displacement by measuring the Euclidian distance between the test and control points (Bayr, 2021). Finally, we measured the distance and angle of incidence between all camera locations and their associated test points to evaluate their influence on point displacement (Stockdale et al., 2015). A Generalized Linear Model was constructed in ArcGIS Pro 2.8.3, using the following equation:

$$\text{point displacement (m)} = \text{intercept} + \beta_1(\text{distance to camera}) + \beta_2(\text{angle of incidence})$$

To satisfy model assumptions, the mean displacement values for each image were fit to a normal distribution using a square root transformation.

We compared the oblique landcover map with landcover data produced from the 30 cm RGBI high-resolution aerial imagery using a Support Vector Machine (SVM) classifier in ArcGIS Pro 2.8.3 (ESRI, 2021). SVMs are supervised machine learning algorithms optimized for locating exemplars (i.e., support vectors) that form decision boundaries for separating landcover classes (Pal & Mather, 2006). The SVM classifier was trained with samples of land cover from the aerial imagery using the same classification scheme used for producing the oblique land cover map (excluding snow/ice and regenerating areas). A total of 273 samples were identified and their distribution across the 6 land cover classes are reported in Table 2.1. The results were resampled to 2m using bilinear interpolation (Appendix D).

We compared accuracy of the landcover maps generated from the oblique and orthogonal imagery based on reference data derived from the 30 cm aerial imagery. We generated 198 equalized stratified random points (i.e., randomly distributed points within each class, in which each class has the same number of points to mitigate class size imbalances) in ArcGIS Pro 2.8.3 and extracted the landcover values from the two maps at each point (ESRI, 2021). We interpreted reference landcover values for each random point by visually assessing the aerial imagery. We then used these data to compute a confusion matrix for each landcover map, which showed the overall accuracy based on the percentage of correctly classified samples. User's accuracy (i.e., the probability that features on the map are present on the ground) and producer's accuracy (i.e., the probability that features on the ground are correctly shown on the map) were reported for each class.

Results

The RMSE for the subset of 7 individual images used to assess georeferencing accuracy ranged from 2.8 to 7.7 m, with an overall mean of 4.6m (Table 2.2). The mean displacement between test and control points ranged from 1.8 to 6.9 m, with a mean of 3.7 m (SD = 2.2 m; Table 2.2). The distance from camera to test points ranged from 1647.7 m to 10 512.9 m, with a mean of 5832.6 m; and the angle of incidence ranged from 5.9° to 23.2°, with a mean of 13.6°. Based on the results of the GLM, there was no significant effect of distance to camera ($P = 0.84$, $\alpha = 0.05$) and angle of incidence ($P = 0.29$, $\alpha = 0.05$) on the mean point displacement.

Table 2.2 Mean georeferencing error (i.e., point displacement) based on 8 test points per image, and mean distance of points from camera and angle of incidence.

Station/Image#	Root mean square error (m)	Mean point displacement (m) (±SD)	Mean distance (m) (range)	AOI (°) (range)
Hawk DSCF0614	3.8	1.8 (±0.6)	4785.5 (3936.3–5955.7)	19.2 (14.9–23.2)
Pyramid DSCF0782	2.8	2.4 (±1.4)	7426.9 (6766.5–8206.9)	13.7 (12.3–15)
Esplanade DSCF0663	4.0	3.5 (±2.1)	5934.4 (4241–8244.4)	13.2 (8.9–17.6)
Palisade DSCF0838	4.6	3.8 (±2.8)	5616.2 (4490–7087)	12.1 (9.4–14.8)
Morro I DSCF0575	4.7	4.1 (±2.2)	2714.5 (1647.7–3267.4)	15.1 (11.9–24)
Greenock DSCF0547	3.6	3.2 (±2)	6983.2 (4263–10512.9)	9.8 (5.9–14.9)
Bonhomme I DSCF725	7.7	6.9 (±4.1)	7367.5 (5388.9–9561.4)	11.7 (7.1–16)
Mean	4.6	3.7 (±2.2)	5832.6	13.6

The two landcover maps are largely consistent in depicting the dominant features of the landscape. Both maps were proficient in classifying the Athabasca River and other water bodies, the open swaths of herbaceous/shrub in the center of the study area, and patches of broadleaf/mixedwood in the northern extent (Fig. 2.4). However, the grain of the two maps is noticeably different owing to the different approaches used. The oblique land cover map is smoother in appearance and presents a more generalized view of the study area. Comparatively, the orthogonal land cover map has the classic speckled appearance of a pixel-based classification and captures more detail on the landscape. This is evidenced by the completeness of the water bodies and linear features, and an overall trend of intermixed classes, specifically where herbaceous/shrub intermixes with barren ground and conifer. However, the latter approach incorrectly classified considerably more broadleaf/mixedwood in the study area because of class confusion with coniferous forest. Wetland was also overclassified consistently throughout the study due to confusion with coniferous forest, herbaceous/shrub, and barren ground.

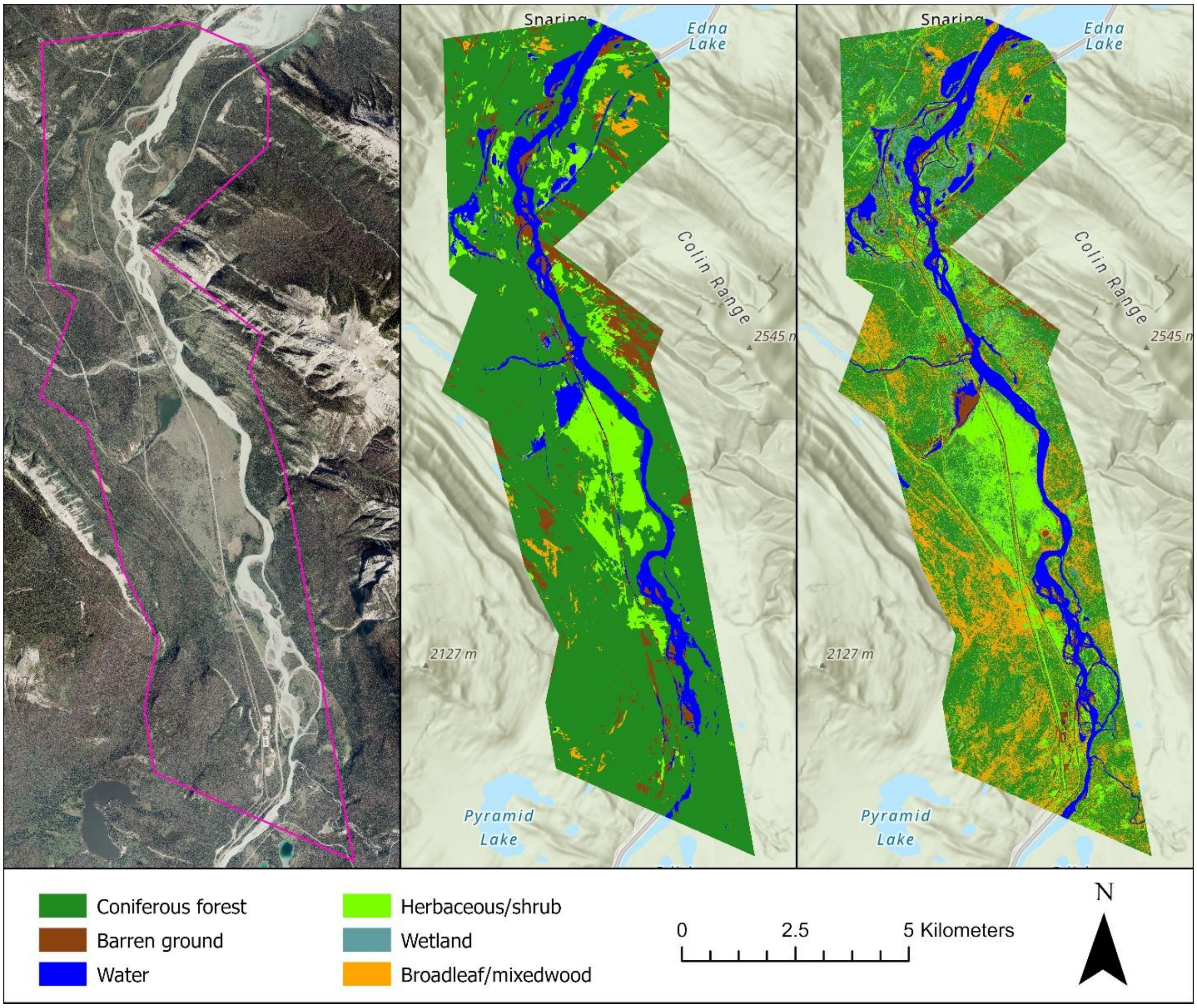


Figure 2.4 Landcover maps produced for the Athabasca River valley study area. Left panel: 30 cm high-resolution aerial image. Middle panel: landcover map produced using oblique photographs (2m resolution). Right panel: landcover map produced using orthogonal imagery (30 cm resolution). Map credit: ESRI, 2021.

In terms of landcover class distribution, barren ground and water shared similar proportions for the two landcover maps while there were considerable differences in the proportions of the other four landcover classes (Fig. 2.5). The biggest difference was for coniferous forest: the oblique landcover map classified 72.3% of the study area as coniferous forest versus 45.2% for the orthogonal map. However, the orthogonal landcover map classified substantially more broadleaf/mixed wood (16.3%) than the oblique map (1.5%). The orthogonal landcover map also classified more wetland (7.9%) and herbaceous/shrub (5.7%) than the oblique landcover map.

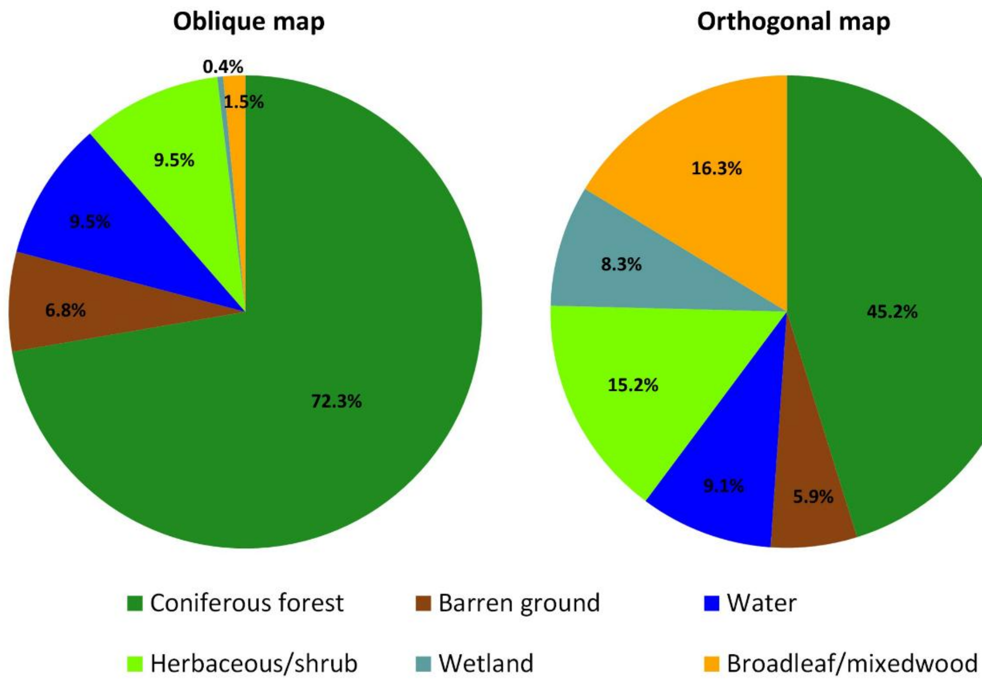


Figure 2.5 Proportion of landcover classes (%) for the oblique and orthogonal maps.

The overall accuracy of the oblique landcover map was 68% (Table 2.3). User's accuracy produced scores above 75% for four of the map classes. However, barren ground (51%) and wetland (33%) had low user accuracy, with the latter class typically misclassified as either herbaceous/shrub or barren ground. Scores for producer's accuracy were more erratic: broadleaf/mixed wood (90%) and water (90%) scored highest; the lowest classes were herbaceous/shrub (55%) and barren ground (49%). The broadleaf/mixed wood and water classes achieved high scores for both producer's and user's accuracy. User's accuracy for the conifer class was 79%, but producer's accuracy was lower due to class confusion with barren ground. There was a larger contrast in scores for herbaceous/shrub, with a high user's accuracy (79%) whereas producer's accuracy (55%) was affected by class confusion with wetland and barren ground.

Table 2.3 Confusion matrix for the oblique landcover map compared to reference aerial imagery indicating overall accuracy of 68%.

Class	Conifer forest	Barren ground	Water	Herbaceous/ shrub	Wetland	Broadleaf/ mixedwood	Total	U accuracy
Coniferous forest	26	2	0	3	1	1	33	0.79
Barren ground	6	17	0	5	3	2	33	0.51
Water	0	5	27	1	0	0	33	0.82
Herbaceous/ shrub	0	3	0	26	4	0	33	0.79
Wetland	1	8	3	10	11	0	33	0.33
Broadleaf/mixed	3	0	0	2	0	28	33	0.85
Total	36	35	30	47	19	31	198	
P Accuracy	0.72	0.49	0.90	0.55	0.58	0.90		0.68

The orthogonal landcover map had an overall accuracy of 60% (Table 2.4) based on the same accuracy assessment points generated for the map derived from oblique photographs. Water (85%) had the highest user's accuracy score, and wetland (38%) had the lowest. Notably, conifer (48%) was frequently misclassified as broadleaf/mixed wood. Similarly, for producer's accuracy, water (93%) scored highest, and wetland (29%) was lowest.

Table 2.4 Confusion matrix for the orthogonal landcover map compared to reference aerial imagery indicating overall accuracy of 60%.

Class	Conifer forest	Barren ground	Water	Herbaceous/shrub	Wetland	Broadleaf/mixedwood	Total	U accuracy
Coniferous forest	25	2	0	5	3	17	52	0.48
Barren ground	1	18	2	2	4	0	27	0.67
Water	0	0	28	0	5	0	33	0.85
Herbaceous/shrub	5	15	0	27	0	1	48	0.56
Wetland	2	1	0	4	5	1	13	0.38
Broadleaf/mixed	3	0	0	6	0	16	25	0.64
Total	36	36	30	44	17	35	198	
P Accuracy	0.69	0.5	0.93	0.61	0.29	0.46		0.60

Discussion

The results of the accuracy assessment indicate the strong potential of a new workflow for oblique photography analysis. When compared to studies georeferencing MLP images with the WSL monoplotting tool, the mean displacement error (3.7 m) for this analysis was considerably lower than Stockdale et al. (2015) and McCaffrey and Hopkinson (2017) who found error values of 14.7 and 21.7 m, respectively. A more recent WSL monoplotting assessment by Bayr (2021) did produce a lower mean displacement error value (1.52 m), but the authors used images taken from lower elevations that were closer to features of interest on the landscape compared to the images in the current study. The steps for georeferencing an image using the IAT georeferencing tool are arguably more intuitive than the WSL monoplotting tool, as (1) fewer control points are required, (2) the reference data (i.e., the virtual photo) shares the same oblique view as the image to be georeferenced, which makes selection of reciprocal control points more intuitive, and (3) the sweep function allows for immediate visual assessment of alignment accuracy. However, there are drawbacks related to the requirement of precise camera station metadata to accurately plot the image classifications. Imprecise camera azimuth and/or coordinates require time-consuming adjustments to produce the optimal virtual photograph for georeferencing. There are opportunities to further develop the IAT georeferencing tool to streamline the adjustment process for existing repeat photograph pairs. New repeat photography fieldwork can also address this issue by using precision instruments to accurately record camera metadata such as survey-grade transits (incorporated into the camera head) and sub-meter GPS units that utilize correction services (Walter, 2020).

The oblique landcover map produced from the oblique images successfully depicted the dominant features of the study area despite a moderate overall score in the accuracy assessment. While several classes produced high scores for both producer's and user's accuracy, the overall accuracy score was impacted by low scores for two classes: wetland and barren ground. Wetland is a challenging class to categorize, for both oblique and orthogonal imagery, given its similarities to other non-wetland classes (Mahdavi et al., 2018). This issue is reflected in class confusion with herbaceous/shrub in the results. Classification accuracy for barren ground was hampered by confusion with numerous classes, especially conifer and wetland. Confusion with the conifer class may be explained by "shadow effect" where patches of barren ground between trees are obscured due to the oblique angle of the photographs. There was also a significant outbreak of mountain pine beetle in the park over the last decade, and the reddish-brown color of beetle-killed pine trees may also have been misclassified as barren ground. A final issue arises from including roads and campgrounds in the barren ground class—these features were often misclassified as wetland as described above.

Comparatively, the overall accuracy for the orthogonal landcover map was lower than the oblique landcover map. The orthogonal landcover map did produce the highest score for an individual class (water), and the SVM algorithm produced better results when classifying barren ground. However, the algorithm was inconsistent when classifying conifer due to a high amount of class confusion with broadleaf/mixed wood forest. This may relate to coniferous trees at different stages of growth (i.e., more open or patchy) confusing the algorithm (whereas this would be less of an issue when viewing coniferous stands at an oblique angle). Similarly, there was considerable class confusion between herbaceous/shrub and barren ground, which may relate to the many areas within the study area where these two classes are intermixed.

A key advantage of the workflow described in this paper is the speed, efficiency, and consistency of classifying images using PyLC. However, noticeable errors are present in the image classifications and the results indicate that PyLC often confused wetland and barren ground with other classes. These errors and misclassifications could be explained by the data used to train PyLC (although georeferencing error may contribute to classification error too). At present the manually classified training datasets constitute less than 0.1% of the complete MLP image collection, and they do not represent the full variety landcover found throughout the region. For example, 50% of the training data is from the Willmore Wilderness Park in western Alberta, where there is a dearth of anthropogenic features. This bottleneck of representative training data limits model sensitivity or classifying additional landcover classes—a more robust model requires a wider range of photographs/classifications to be included in the training data.

Improving PyLC accuracy could be addressed in several ways. Manually corrected PyLC classifications are considerably quicker to produce than full manual classifications and could be an effective approach to expanding training datasets (especially for underrepresented ecoregions in the Canadian cordillera). Corrections typically require 1–3 h of work per image and can be performed using photo-editing software applications. Additionally, further work to refine data augmentation methods can potentially address a severe class imbalance in the training data sets, as can models optimized for classifying specific environments. The addition of new landcover classes (i.e., developed areas/infrastructure, mountain pine beetle impacted forest, etc.) may improve the classification accuracy for other classes. Finally, PyLC improvements that incorporate semi-supervised approaches to segmentation have the potential to reduce reliance on labeled training data (Hong et al., 2015; Hung et al., 2018). Further, unsupervised segmentation approaches employing iterative self-training procedures have potential to eliminate the need for training data altogether (Zou et al., 2018).

The mosaicking method for combining the 19 georeferenced image classifications makes it challenging to track the source of error in the oblique landcover map. However, this approach can correct misclassification errors from individual images when they are mosaicked together using the mode value technique. An example demonstrating the advantage of this method is found at a lake located to the south of the Snaring River mouth (Fig. 2.6). One of the classified images from the Hawk Mountain station incorrectly classifies the lake as conifer, whereas this error is corrected in the combined oblique landcover map. Another advantage of this approach is the mitigation of the shadow effect of the oblique photographs, whereby landcover in the viewshed is obscured by trees or elevated terrain. Combining georeferenced images from different angles in the study area can fill in these gaps and address issues where the height of trees from the oblique angle does not result in an exaggerated footprint on the landscape.

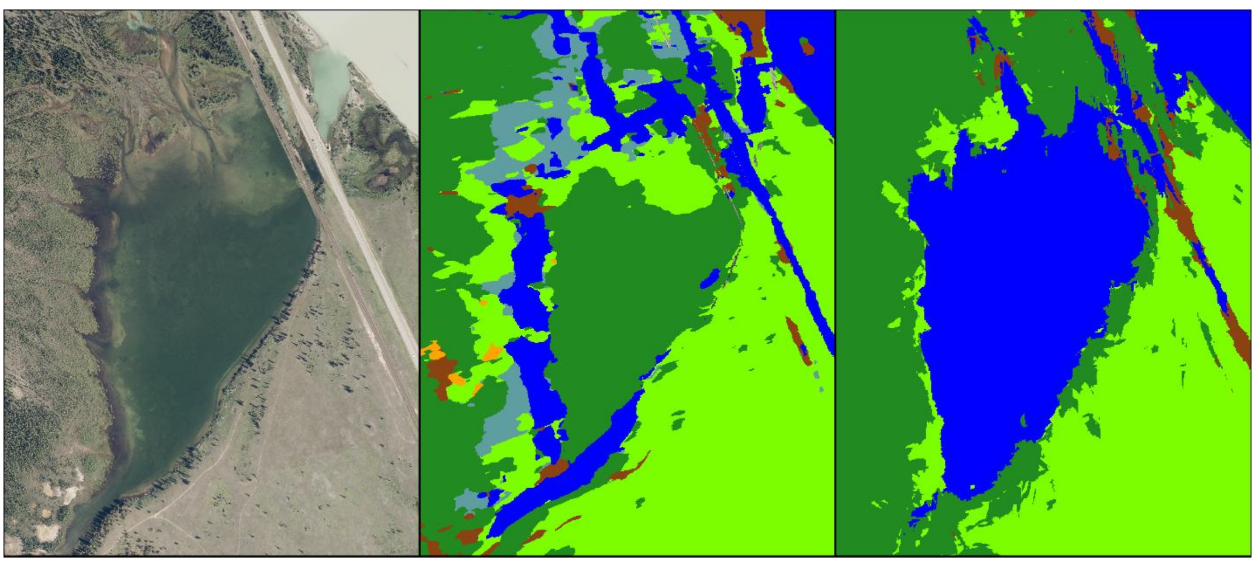


Figure 2.6 Example of a classification error in an individual georeferenced image classification that was corrected when mosaicking all georeferenced image classifications together using the mode value technique. Left panel: orthophoto showing the extent of the lake. Middle panel: the lake is misclassified as conifer in single georeferenced image classification (derived from a photograph captured from the Hawk Mountain station). Right panel: the lake is correctly classified as water in the oblique landcover map comprising 19 georeferenced image classifications.

Other important considerations for the workflow detailed in this paper relate to image quality, DEM resolution, and the terrain of prospective study areas. While PyLC is capable of classifying images of varying quality (i.e., images captured with sub-optimal lighting), some images do not lend themselves well to classification due to the elevation of camera station relative to the landscape below, foreground vegetation obscuring large parts of the field of view, or simply the distance from the camera to the landcover area of interest. The accuracy of the georeferenced image classifications is largely related to the resolution of the DEM employed by the IAT georeferencing tool. To date, there is limited coverage of fine- resolution LiDAR-derived DEM data in the Canadian Rockies and other mountainous regions in western Canada. The availability of high-resolution elevation data should be considered when undertaking such work for a specific area. Another consideration is the elevation of prospective study sites. The analysis presented in this paper is focused on subalpine valley ecosystems, but the real potential of these methods may lie in alpine environments where the oblique angle of the photographs can offer richer detail than orthogonal imagery (Fortin et al., 2019). This would be especially true for ecotone shift assessments (i.e., McCaffrey & Hopkinson, 2020; Peterson et al., 2022; Trant et al., 2020).

A logical next step is applying the workflow presented in this paper to historical survey photographs in the MLP collection captured between 1880 and 1950. This information would predate the first aerial imagery surveys by decades and would deepen existing monitoring reference data, providing insights into the configuration and composition of ecosystems under historically contingent disturbance regimes (Keane et al., 2009). Quantifying the historical range of variability for ecosystems (i.e., Landres et al., 1999) can provide a spatially explicit baseline for evaluating future responses to altered disturbance regimes, climate warming, and other anthropogenic stressors (Turner & Seidl, 2023). However, different machine learning models are required to classify color (3-band) and grayscale (single band) images. While PyLC models configured for classifying historical photographs have been developed (Rose, 2020), the image classifications produced from historical grayscale photographs presently lack the accuracy and consistency of the classifications produced from their color counterparts. Work is underway to improve machine learning approaches to historical imagery.

In conclusion, the new workflow described in this paper advances the use of oblique photographs to yield quantitative landcover data. Our approach has several advantages over existing methods, specifically the ability to produce quick and consistent image classifications with little human input, and accurately georeference and combine these classifications to generate landcover data for large areas. We have demonstrated that this approach produces results of comparable accuracy to landcover maps generated using supervised classification and orthogonal imagery. Further developments for improving this workflow would be best focused on three areas: (1) expanding training data for PyLC that includes new landcover classes; (2) improving the performance of PyLC algorithm (i.e., incorporating updates to Deeplabv3+); and (3) increasing the accuracy of camera station metadata to expedite georeferencing tasks. Additionally, new research is required to investigate the potential of this new workflow to support management practices in protected areas. Historical and repeat landcover maps derived from oblique photographs can provide valuable data for vegetation monitoring, restoration activities, and management planning in an era of rapid change.

Acknowledgments

We thank Mary Sanseverino and Michael Whitney for their important and ongoing work in developing the IAT georeferencing tool, and their thoughtful reviews that greatly improved this manuscript. We thank Sonia Voicescu, Kristyn Lang, and Mike Eder for their assistance in collecting field data. And we thank Jasper National Park, fRI Research, Mitacs Accelerate, Lorene Kennedy Environmental Studies PhD Award, Lorene Kennedy Field Research Award, and SSHRC for supporting this research.

Chapter 3: A tale of two disturbances: can mountain pine beetle restore landcover composition and pattern altered by fire suppression in Jasper National Park?

Abstract

Disturbance is a key driver of heterogeneity in mountain landscapes. Historically, the montane ecoregion (i.e., lower elevations where the climate is typically warmer and drier) in Jasper National Park experienced a mixed-severity fire regime with short fire return intervals. However, fire suppression over the previous century has altered the fire return interval, resulting in a significant increase of homogenous forest cover. These changes contributed to an extensive mountain pine beetle (MPB; *Dendroctonus ponderosae*) outbreak in the park over the previous decade, which impacted up to 70% of lodgepole pine (*Pinus contorta*). With no previous records of MPB in the park, new research is required to investigate whether the composition and pattern of ecosystems in the montane ecoregion have departed from their historical range of variability due to altered or novel disturbance. In this study, we compare the composition and pattern of landcover in the aftermath of historical, stand-replacing fires (1889 and 1905) and the recent MPB outbreak. We repeated historical survey photographs (captured in 1915) and employed a recently developed workflow for classifying and georeferencing oblique images to produce historical and contemporary landcover maps for a study area in the montane ecoregion. Our results indicate that the MPB outbreak has reduced the area-weighted mean patch size for mature conifer and increased patch complexity in our study area. We conclude that MPB has reintroduced heterogeneity to the montane ecoregion, but also acknowledge that succession in MPB-affected stands will likely differ from post-fire disturbance as the insect only affects pine species. The timing of new compounding disturbances will further influence the recovery and future composition of species in the study area.

Introduction

At the turn of the twentieth century, the central valleys of Jasper National Park (hereafter "Jasper NP") in Alberta, Canada, were recovering from pervasive disturbance. Dendrochronological records indicate large, moderate-to-high severity wildfires burned through the area in 1889 and 1905 (Chavardès and Daniels 2016; Tande 1979). Consequently, this montane ecosystem was characterized by a patchy mosaic of regenerating forest and shrub, interspersed with open, mature forest, and myriad lakes, wetlands, and rivers. This landcover mosaic is evident in hundreds of photographs captured from survey stations above the valleys by the Dominion Lands Survey (DLS) in 1915 (Fig. 3.1). Eighty years later, Rhemtulla et al. (2002) repeated the original DLS survey photographs and developed a quantitative method to assess vegetation change between the image pairs. The results indicated a significant increase in forest cover dominated by coniferous species (Fig. 3.1). The vegetation changes were attributed to a decrease in fire frequency and the authors forewarned that Jasper NP's maturing, even-aged forest stands were susceptible to future insect disturbance. Indeed, over the next decade, an outbreak of mountain pine beetle (MPB; *Dendroctonus ponderosae* Hopkins) erupted in the park, impacting up to 70% of lodgepole pine (*Pinus contorta*) and rendering large swaths of forest with a telltale reddish hue (Fig. 3.1).

Disturbance is a key driver of ecosystem dynamics and can be defined as "any relatively discrete event that disrupts the structure of an ecosystem, community, or population, and changes resource availability or the physical environment" (White and Pickett 1985). This disruption changes the trajectory of an ecosystem and is fundamental in maintaining spatial and temporal heterogeneity evident at broader scales (Turner 2010). Natural disturbances occur over a range of time periods, and may be abiotic or biotic in origin, or a combination of the two (Turner 2010). A disturbance regime refers to the cumulative effects of a reoccurring disturbance agent over space and time (Keane 2013). Climate has a strong influence on many disturbance regimes, affecting their intensity, severity, frequency, extent, and duration (Seidl et al. 2020). Over time, the bounds of ecosystem variability become defined as species adapt to reoccurring disturbance. The concept historical range of variability characterizes the conditions between these bounds, providing a sense of the range of ecosystem characteristics exhibited in response to disturbance and recovery over time and space (Keane et al. 2009; Landres et al. 1999; Morgan et al. 1994).

Figure 3.1 Top: Historical DLS survey photograph captured in 1915 from the Power House Cliff survey station in Jasper NP. Middle: Repeat photograph captured in 1998. Bottom: Second repeat photograph captured in 2019. Source: Mountain Legacy Project https://explore.mountainlegacy.ca/stations/show/812

In the Canadian Rockies, fire is recognized as a primary disturbance agent influencing forest dynamics, but insects, pathogens, floods, avalanches, and windstorms also play a role (Luckman 1998; Wright and Heinselman 1973). Wildfires occur both via lightning strikes and anthropogenic sources. Historically, Indigenous Peoples have used fire for a variety of uses, including to maintain travel routes and create forage to attract wildlife, and there is some evidence that the historic fire regime in Jasper NP was fueled primarily by human ignition (Lewis and Ferguson 1988; Murphy 2007; Turner 1999). Lodgepole pine (*Pinus contorta* var. *latifolia*), the dominant coniferous species in the montane ecoregion, is a fire-adapted/dependant species (Brown 1975). Mixed-severity and stand-replacing fires support spatial complexity and diverse species and age groups across the landscape.

However, montane ecosystems in the Canadian Rockies may have departed from their historical range of variability (HRV) due to the influence of altered and novel disturbance regimes. Fire return intervals (i.e., time between fires occurring on a landscape) have increased due to the effectiveness of pervasive fire exclusion (Keane et al. 2002). As visitation surged in the Canadian Rocky Mountain Parks after the Second World War, fire was viewed as a threat to wilderness values (Woodley 1995). Park managers made use of new equipment, constructed fire roads, and installed fire tower networks to greatly improve fire suppression responses over the ensuing decades (Keane et al. 2002; Woodley 1995). Consequently, the buildup of hazardous fuels ultimately made it more difficult to fight fires. This issue has been exacerbated by climate warming which has lengthened fire seasons, decreased fuel moisture content, and increased natural ignitions, fire spread rates, fuel availability, and flammability (Seidl et al. 2017).

Altered fire regimes and climate warming in the Canadian Rockies have also created ideal conditions for MPB population growth. Native to western North American forests, MPB are innocuous in low numbers but under favorable conditions they can erupt to epidemic levels and attack mature pine species over vast areas (Safranyik and Carroll 2006). MPB outbreaks have occurred in mountain National Parks in the southern Canadian Rockies over the previous century but there are no records of the insect in Jasper NP prior to 1999 (Dalman 2004). However, a century of fire exclusion has resulted in a buildup of mature even-aged stands of lodgepole pine throughout the park's montane ecoregion. Additionally, the cold winter temperatures in the Canadian Rockies that limited the range of MPB have become more erratic with milder winters increasing overwintering survival of MPB broods (Cooke and Carroll 2017). These factors (abundance of host trees and mild winter temperatures) were the catalyst for a massive MPB outbreak beginning in central British Columbia in the late 1990s. Through long-range dispersal, the insect rapidly expanded beyond their historical range into central and northern Alberta, where evolutionary naïve pine forests are more susceptible to MPB impacts (Burke et al. 2017; Cudmore et al. 2010; de lay Giroday et al. 2012; Raffa et al. 2008; Robinson 2015). The expansion of MPB into new habitats where they have not been present historically is an example of novel disturbance (Lieffers et al. 2023; Turner and Seidl 2023).

Altered and novel disturbances can drive changes in ecosystem composition and configuration, resulting in departures from their HRV. In Jasper NP, the historical and repeated survey photographs depict montane landcover in the aftermath of two extensive disturbance events. However, the recent MPB outbreak in the park is historically unprecedented and raises important questions about whether the characteristics of this event are within the HRV of the montane ecosystems. The focus of this study is to compare the composition and pattern of the montane ecosystems after the two disturbance events to better understand what impact the novel insect disturbance imparted on landcover in Jasper NP's montane ecoregion. To undertake this comparison, we employ a recently developed workflow for classifying and georeferencing oblique images to produce historical (1915) and contemporary (2020–2022) landcover maps for a study area in Jasper NP (Tricker et al. 2024). We use these two maps to assess the landcover changes that have occurred in the park since fire suppression began. We then overlay MPB extent data onto the contemporary map to assess the degree to which the MPB outbreak might be analogous to historic fire. We use this analysis to consider whether novel insect disturbance can reintroduce landcover composition and pattern altered by fire suppression in Jasper NP's montane ecoregion.

Study area

The study area comprises 26,424 hectares of the Miette and Athabasca River valleys in Jasper NP, located in the eastern Canadian Rockies in west-central Alberta (Fig. 3.2). This constitutes much of the lower elevation region of the park, where major national transportation routes and the town of Jasper are located. The park was established in 1907 and extends over 11,000 km^2. The steep elevational gradient in the park has a strong influence on the local climate and vegetation, resulting in three distinct ecoregions (Holland and Coen 1983). The lowest elevations comprise the montane ecoregion (1000–1350 m), where the climate is typically drier and warmer (Holland and Coen 1983). Lodgepole pine is the dominant tree species, although Douglas Fir (*Pseudotsuga menziesii*), white spruce (*Picea glauca*), and trembling aspen (*Populus tremuloides*) are also present (Holland and Coen 1983). Grasslands are interspersed among woodlands in the montane ecoregion, and wetlands are often found adjacent to water bodies (Beschta and Ripple 2007). The subalpine (1350–2300 m) and alpine (> 2300 m) ecoregions occupy the higher elevations in the park, where the climate is colder and wetter (Holland and Coen 1983). The mid-elevation subalpine ecoregion typically comprises subalpine fir (*Abies lasiocarpa*), Engelmann spruce (*Picea engelmanii*), and lodgepole pine forests, transitioning into open shrub-forb meadows in the high-elevation alpine ecoregion (Bradley and Neufeld 2012).

Dendrochronological analyses in the montane forests (that overlap with the present study area) indicate that historically Jasper NP experienced a mixed-severity fire regime with short fire return intervals. A study by Tande (1979) examined a 43,200 ha area centered on the town of Jasper and found evidence of 46 fires that burned between 1665 and 1913 (mean fire return interval of 5.5 years), most of which were low to medium intensity. A more recent study by Chavardès and Daniels (2016) assessed a 2000 ha area located 12 km north of the town of Jasper, revealing evidence of 13 mixed-severity fires between 1646 and 1905 (mean fire return interval of 20 years). Both studies found evidence of widespread (i.e., > 50% of study area), medium to high intensity fires, which had a mean fire return interval of approximately 65 years. The most recent of these widespread fires occurred in 1889 (Tande 1979) and 1905 (Chavardès and Daniels 2016), with the former affecting up to 21% of the total area of Jasper NP (Luckman 1998). Both studies confirm that the fire regime and successional dynamics in the montane ecoregion have been substantially altered in the twentieth century due to fire exclusion.

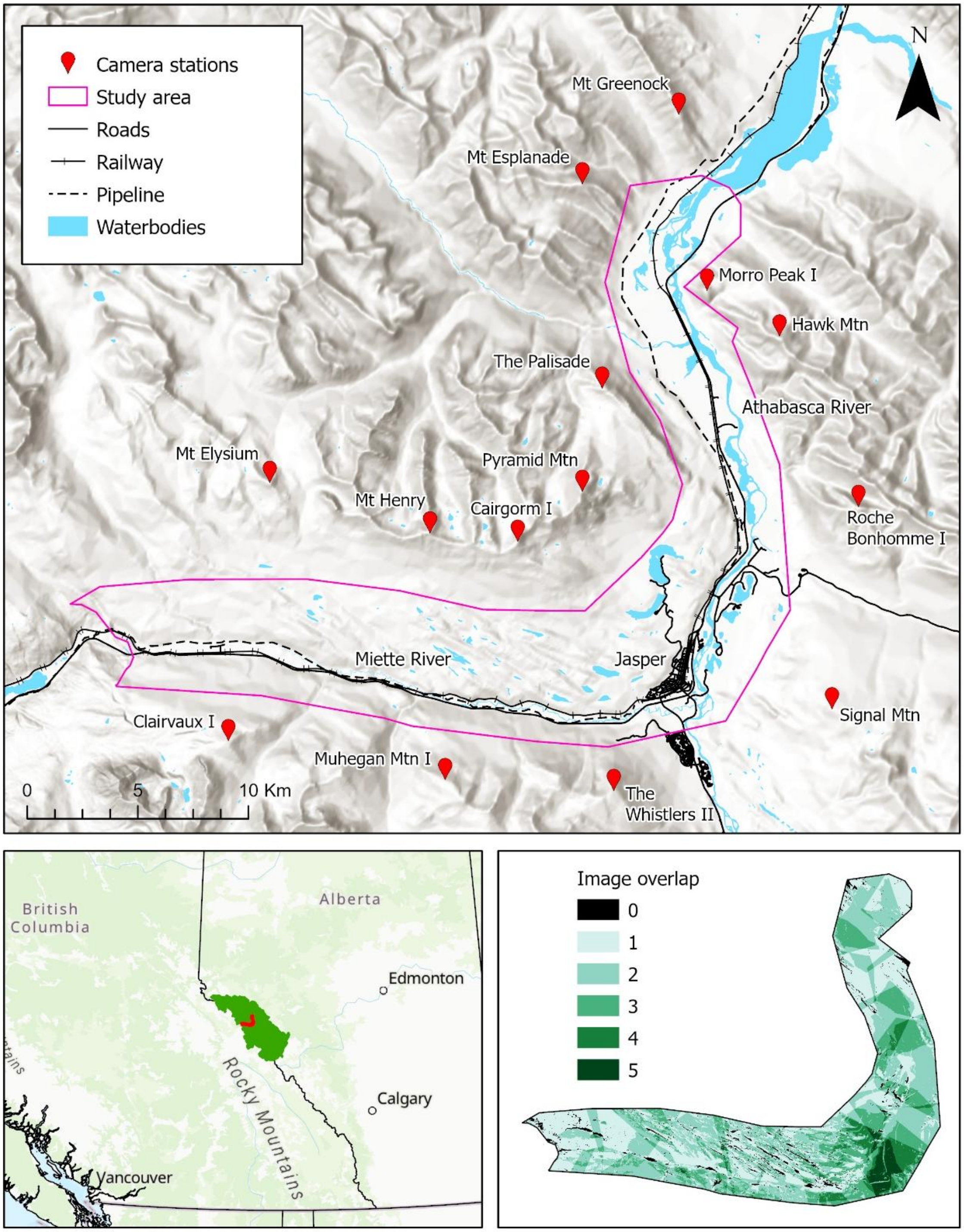


Figure 3.2 Map of the study area (top panel) in Jasper National Park showing the survey stations (red pins) from which historical and repeat photographs were taken. The town of Jasper is located in the middle of the study area. Bottom left panel shows the location of the study area (red) in Jasper National Park (green) in Western Canada. Bottom right panel shows number of overlapping images across the study area.

Methods

We use a new workflow to classify and georeference historical and contemporary oblique photographs to produce 1915 and 2020/22 landcover maps for the study area in Jasper NP. We evaluate changes in the composition and pattern of the two maps that contributed to the extensive MPB outbreak that began in the park around 2010. We then assess the impact of the MPB outbreak on landcover composition and pattern in the 2020/22 map and compare these changes to the landcover depicted in the 1915 map.

The workflow for producing landcover data from oblique photographs was developed by researchers associated with the Mountain Legacy Project (https://mountainlegacy.ca/; Tricker et al. 2024). It incorporates two tools for classifying and georeferencing oblique images. The Python Landscape Classifier (PyLC; https://github.com/scrose/pylc) is a deep learning segmentation network trained on historical and repeat MLP images and their corresponding landcover classifications (Fortin et al. 2018; Jean et al. 2015; Rose 2020). PyLC classifies images into 9 landcover classes, which are based on the broad habitat types found in the Canadian Rockies (Table 3.1). The Image Analysis Toolkit (IAT; Sanseverino et al. 2016) georeferencing tool uses a web-based ray tracing algorithm to relate an image's field of view (FOV) to its geographic location using camera metadata (i.e., the camera location, azimuth, and FOV width) and a digital elevation model. Tricker et al. (2024) recently tested the accuracy of a landcover map produced from 19 oblique images using the workflow, reporting an average root-mean square error of 4.6 m for georeferencing accuracy and 68% overall classification accuracy for the landcover map.

Table 3.1 Description of landcover classes used for classifying the historical and repeat images.

Landcover classes	Description
Mature conifer	Greater than 75% mature coniferous trees
Broadleaf/mixed wood forest	> 70% broadleaf trees/patches with 30–70% broadleaf cover and the rest coniferous trees and/or shrub
Regenerating conifer	Young coniferous forest which is distinguishable from taller mature coniferous forest
Recently burned	Visibly recently burned forest
Shrub	> 25% shrub cover
Herbaceous	Forb-grassland < 25% shrub cover
Wetland	Vegetation with a wet or aquatic moisture regime
Water	Lakes and rivers
Barren ground	Soil, sand, gravel, rock, or anthropogenic impervious surfaces

The historical survey photographs used in this analysis were captured by a DLS team led by Morrison Bridgland in 1915. The surveyors captured a total of 735 photographs on 4 x 6 inch glass plates from a network of 92 survey stations located predominantly on peaks and promontories above the three major valleys in the park. The glass plates were eventually stored at Libraries and Archives Canada (LAC) after Bridgland had used the images and theodolite measurements to produce topographical maps for the central portion of Jasper NP. To select images for inclusion in our analysis, we produced viewsheds for each image in ArcGIS Pro 3.0.2 (ESRI 2022) using the survey station coordinates and the image azimuths recorded by Rhemtulla et al. (2002) and a composite 2 m digital surface model. We developed digital surface and terrain models (DSM and DTM) for the study area using elevation data from Natural Resources Canada (2021) and airborne LiDAR data obtained from Jasper NP. We identified all images that had a viewshed footprint in the montane ecoregion of the Athabasca and Miette River valleys, and further refined this selection by assessing the quality (i.e., exposure and clarity) of each image. This left a final selection of 26 images captured from 14 survey stations, which provided 97.1% coverage of the study area (Appendix E; Fig. 3.2). We obtained high-resolution scans (1600 dpi) of the historical images from LAC.

We undertook fieldwork in Jasper NP in the summers of 2020 and 2022 to capture repeat photographs of the 1915 images. We used a 51.4-megapixel FujiFilm GFX50s medium format digital camera with a 32–64-mm F4 zoom lens, and a custom-configured Novoflex panoramic head and tripod and followed protocols developed by the MLP (Fortin et al. 2018). This involved using gridded printouts of the original photographs to finetune the tripod position and align each repeat photograph through the camera's electronic viewfinder. We also recorded detailed field notes, including the camera location coordinates with an EOS Arrow 100 submeter GNSS receiver and the image azimuth with a Brunton transit. We visited all 14 survey stations and captured a total of 127 repeat photographs. These images were then uploaded to MLP Explorer—a map-based tool developed by the MLP for storing and viewing digital copies of the historical and repeated survey photographs in the MLP collection (https://explore.mountainlegacy.ca/).

The historical and repeat images were classified into the 9 landcover classes listed in Table 3.1 using PyLC. Manual corrections were undertaken in Affinity Photo Version 2.3.0 to improve classification accuracy for all landcover classes and remove tiling artifacts introduced by PyLC

(Appendix F). We did not classify beetle-killed forest given the difficulty of determining the extent and percent mortality in the repeat images. The 26 classified image pairs were then georeferenced using the IAT georeferencing tool following the steps outlined in Tricker et al. (2024). All classified images were georeferenced at 2 m resolution (i.e., the same resolution as the elevation datasets). To produce the historical (1915) and repeat (2020/22) landcover maps, the overlapping georeferenced classifications were combined using the majority value (i.e., the value that occurs most often) in ArcGIS Pro 3.0.2 with the cell statistics tool (ESRI 2022). This provided coverage for 92.3% of the study area (the remaining area had multiple majority values). To fill in the remaining areas, we followed an approach described by Tricker et al. (2024) involving iteratively dropping individual georeferenced classifications and recalculating the majority value until only one georeferenced classification remained. The resulting majority value grids were then mosaicked together using the Mosaic to New Raster, which brought the overall mapping coverage up to 97.1% (ESRI 2022). Finally, we used the Nibble tool to fill in the remaining gaps based on the nearest neighbor values (ESRI 2022).

We obtained 30-cm RGBI aerial imagery (captured in 2020) from Jasper NP to assess the accuracy of the 2020/22 landcover map. We generated 400 equalized stratified random points in ArcGIS Pro 3.0.2, which accounts for class imbalances by producing the same number of random points for each landcover class (ESRI 2022). We visually interpreted landcover from the aerial imagery at each random point, and constructed a confusion matrix to determine the overall, user's, and producer's accuracy of the 2020/22 landcover map. To assess landcover changes between the 1915 and 2020/22 maps, we calculated the area and percentage for the landcover classes in each map. We then used the Compute Change tool in ArcGIS Pro 3.0.2 to determine the spatial extent of the categorical changes between the two maps, with a focus on classes that converted to mature conifer in the 2020/22 landcover map (ESRI 2022). To determine changes in landcover pattern in the study area, we first used the focal statistics tool in ArcPro 3.0.2 to apply a 10 x 10 cell neighborhood to the maps using the majority statistic to remove mosaicking artifacts, then calculated the area-weighted mean (i.e., each class is weighted by its proportional area in the study area) for the shape index (a measure of patch complexity) and patch size for each landcover class using Fragstats 4.2 (ESRI 2022; McGarigal et al. 2012).

Finally, to determine the amount of mature conifer impacted by MPB in the 2020/22 landcover map, we obtained the 2022 Vegetation Resources Inventory (VRI) dataset from Jasper NP (Government of British Columbia 2022). We used this dataset because PyLC is unable to distinguish live from dead canopy cover. We selected all polygons that were impacted by MPB by querying the disturbance join table. The Disturbance Percent Code for each selected polygon provides an estimate of percent basal area (or volume) of MPB-caused mortality, summarized in 10% bins (Fig. 3.3). We used these data to determine which areas of the 1915 landcover map would go on to be impacted by all MPB mortality bins in the future. We also used the 31–100% bins (i.e., moderate to high MPB severity) to determine which areas of mature conifer would likely undergo changes in the 2020/22 landcover map. We recalculated the area and percentage of the adjusted landcover in ArcGIS Pro 3.0.2 and the area-weighted mean shape index and patch size in Fragstats 4.2 (ESRI 2022; McGarigal et al. 2012).

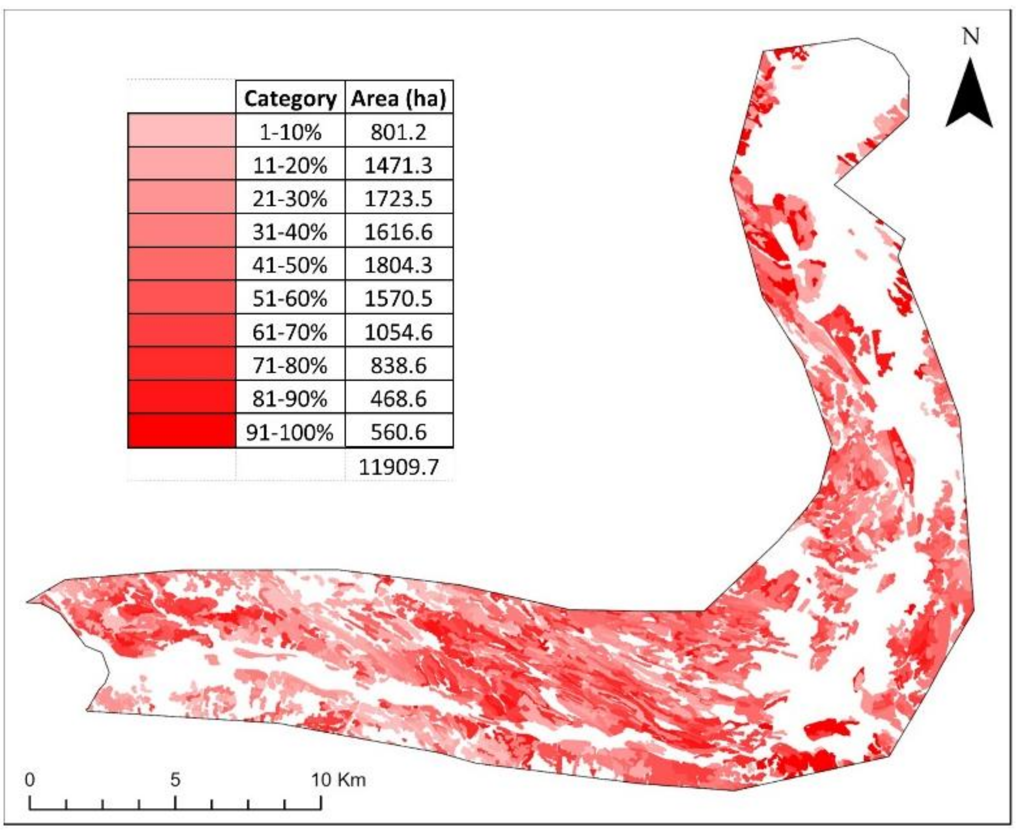


Figure 3.3 Extent of MPB mortality in the Jasper NP study area. Mortality is an estimate of the percent basal area of dead trees in each polygon, summarized into 10% bins. Source: Government of British Columbia 2022.

Results

The predominant landcover class in the 1915 landcover map is regenerating conifer (56.4% or 14,896.7 hectares) and is found throughout the western portion of the study area and along the western flanks of the northern portion (Table 3.2; Fig. 3.4). The second largest class is mature conifer (20.1%), with large stands present to the west of the study area along the Miette River and around Jasper townsite. Large stands of mature conifer are also present east of the Athabasca River in the central portion of the study area, and on the western flanks of the upper northern portion. Smaller stands of mature conifer are found adjacent to the Athabasca River throughout the northern portion. The main areas of shrub (10.4%) are located east of the Athabasca River and the northern corner of the study area. Water, comprising rivers and lakes, makes up 4.8% of the study area. Barren ground (3.4%) comprises sand bars located along river courses, bands of rock in upper elevations of the northern portion of the study area and among the benchlands of the western portion, and roads and railways in the valley bottoms. Most of the herbaceous cover (2.2%) is located to the west of the Athabasca River and in small pockets within the Jasper townsite and toward the northern extent of the study area. This is also where a majority of wetlands (1.8%) are located, along with smaller areas along the Miette toward the western entrance of the park. Broadleaf/mixed wood forest (0.4%) and recently burned areas (0.4%) are the smallest classes, with former found in small stands throughout the study area and the latter located in upper elevations north of the Miette River. Regenerating conifer had the highest area-weighted mean shape index (25.91) and the largest area-weighted mean patch size (10,968.2 ha).

The 2020/22 landcover map is dominated by the mature conifer class (21,854.7 hectares), which now comprises 82.7% of the study area (Table 3.2; Fig. 3.4). Water (4.6%) is the next largest class, followed by barren ground (3.1%). Water is largely found in the same locations as the 1915 landcover map, but barren ground has decreased in upper elevations and gained area in the valley floor due to an increase in anthropogenic features and infrastructure, especially around the town of Jasper (Table 3.2). Shrub has decreased to 2.6% and is mostly found in the northern portion at lower elevations but also on the benchlands northwest and west of Jasper townsite. Stands of broadleaf/mixed wood forest (2.4%) have increased throughout the study area, specifically in areas north, southwest, and southeast of the town, and along the eastern flanks of the northern portion. Herbaceous cover (2.4%) increased slightly despite the large area of grassland decreasing in the northern portion of the study area, with new areas emerging west, north, and east of the town.

Wetland (2.0%) also increased slightly and is found largely in the same areas as the 1915 landcover map. The regenerating conifer class (0.2%) decreased significantly and is only found on the eastern flanks of the northern portion of the study area. There are no recently burned areas in the 2020/22 landcover map. Mature conifer had the highest area-weighted mean shape index (18.01) and the largest area-weighted mean patch size (15,228 ha).

Table 3.2 Area and percentage of landcover classes for the 1915, 2020/22, and MPB-adjusted 2020/20 landcover maps.

Landcover class	1915 map				2020/22 map				MPB-adjusted 2020/22 map			
	Area (ha)	%	Area-weighted mean shape index	Area-weighted mean patch size (ha)	Area (ha)	%	Area-weighted mean shape index	Area-weighted mean patch size (ha)	Area (ha)	%	Area-weighted mean shape index	Area-weighted mean patch size (ha)
Mature conifer	5308.9	20.1	5.62	226.5	21854.7	82.7	18.01	15228.0	14209.1	53.8	23.69	6381.8
Broadleaf/ mixed wood	118.7	0.4	2.11	6.4	637.2	2.4	2.77	17.8	640.9	2.4	2.78	17.8
Regenerating conifer	14896.7	56.4	25.91	10968.2	52.7	0.2	2.95	3.6	52.7	0.2	2.95	3.6
MPB-affected conifer	0.0	0.0	0.00	0.0	0.0	0.0	0.00	0.0	7630.5	28.9	6.66	393.7
Recently burned	100.3	0.4	2.62	14.7	0.0	0.0	0.00	0.0	0.0	0.0	0.00	0.0
Shrub	2748.4	10.4	5.94	178.5	684.0	2.6	3.08	16.8	686.0	2.6	3.08	16.8
Herbaceous	583.6	2.2	5.39	213.1	640.1	2.4	3.77	49.4	642.3	2.4	3.77	49.4
Wetland	482.0	1.8	3.92	32.1	536.8	2.0	3.06	19.8	537.4	2.0	3.07	19.8
Water	1280.0	4.8	7.76	206.7	1206.4	4.6	8.46	202.7	1207.4	4.6	8.46	202.6
Barren ground	905.6	3.4	3.80	14.0	812.6	3.1	5.89	65.8	818.0	3.1	5.90	65.5

The overall accuracy of the 2020/22 landcover map was 79% (Appendix G). Water and herbaceous (both 94%) scored highest for user accuracy (i.e., the probability that features on the map is present on the ground). Mature conifer (80%) saw some confusion with broadleaf/mixed wood forest (74%), and vice versa. The lowest score was for barren ground (64%), which was confused with mature conifer, water, and shrub. The highest scores for producers' accuracy (i.e., the probability that features on the ground are correctly shown on the map) were regenerating conifer (97%), herbaceous (96%), and wetland (91%). Shrub (59%) was regularly confused with regenerating conifer, and mature conifer (65%) was misclassified as broadleaf/mixed wood forest or barren ground.

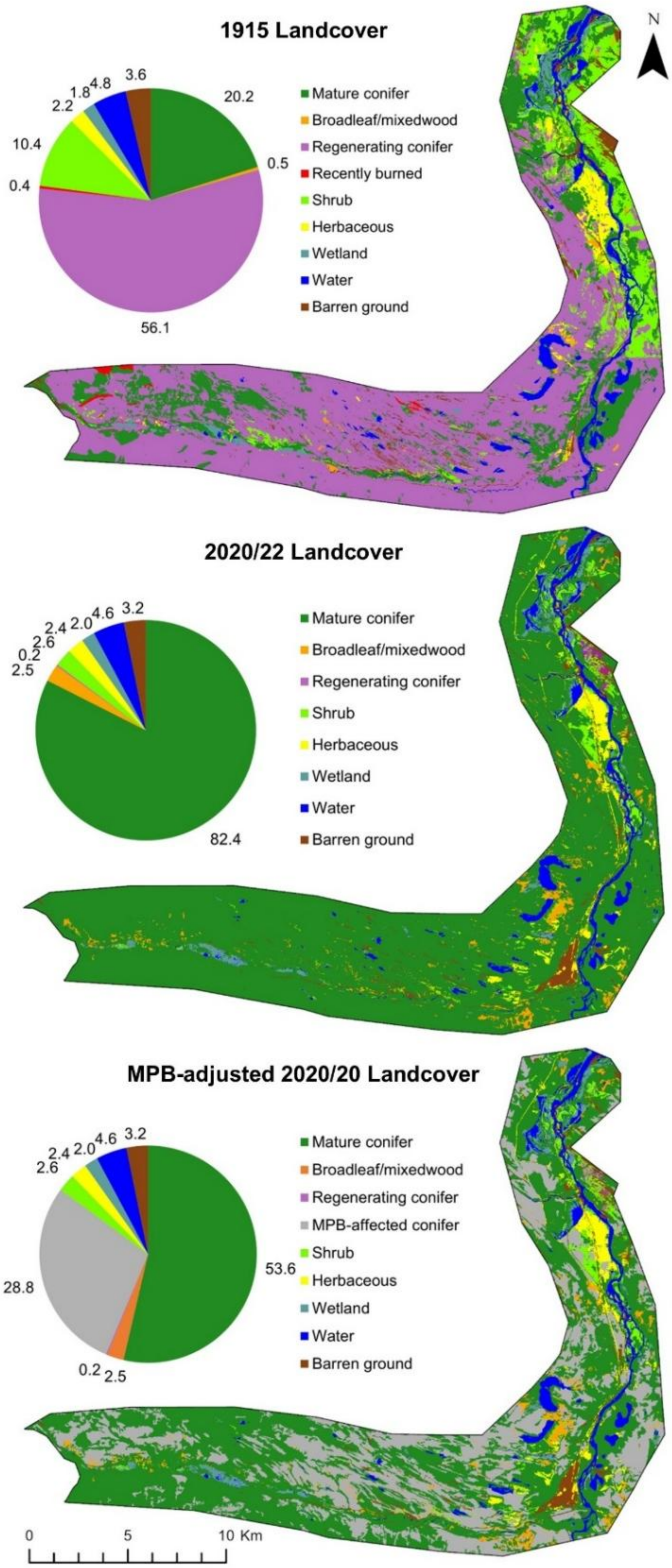


Figure 3.4 The 1915 (top) and 2020/22 (middle) landcover maps derived from the historical and repeat photographs. The MPB-adjusted 2020/22 map (bottom) reflects mature conifer affected by moderate to high MPB severity. The pie charts depict the percent of landcover for each map.

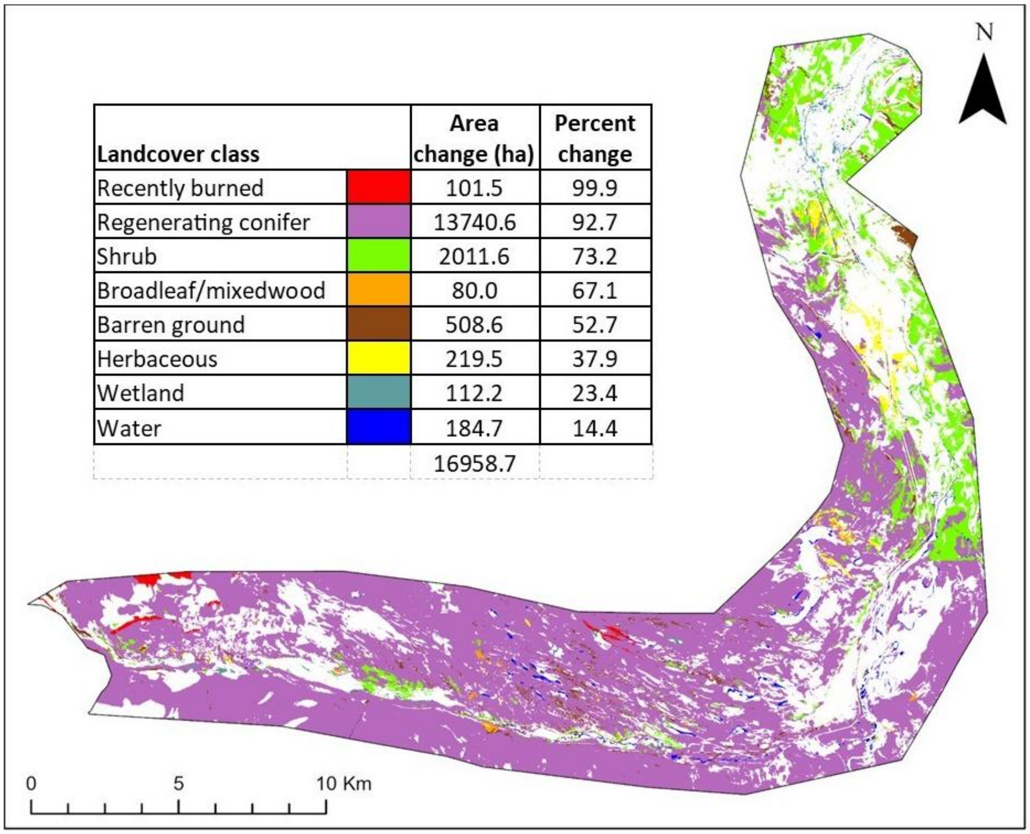
The change analysis showed that a majority of the historical landscape (16,958 ha, 64.2%) transitioned to mature conifer cover in 2020/22 (Table 3.3; Fig. 3.5). Most of the new mature conifer cover originated from regenerating conifer (13,740.6 ha, 92.7%), followed by shrub (2011.6 ha, 73.2%). Almost all regenerating area (101.5 ha, 99.9%) transitioned to mature conifer, followed by barren ground (508.6 ha, 52.7%), broadleaf/mixed wood forest (80 ha, 67.1%), and herbaceous (219.5 ha, 37.9%). Table 3.4 indicates the area (ha) and percent of classes in the 1915 landcover map that would be impacted by MPB (using all mortality bins) a century later. This includes non-forest classes in 1915 that transitioned to forest classes in 2020/22.

Table 3.3 Change matrix tracking differences in landcover classes (ha) from 1915 (left) to 2020/22 (top).

Landcover class	Mature conifer	Broadleaf/ mixed wood	Regenerating conifer	Shrub	Herbaceous	Wetland	Water	Barren ground
Mature conifer	4823.8	49.6	3.3	105.2	89.8	73.1	68.3	114.1
Broadleaf/mixed wood	80.0	29.0		4.9	3.3	0.3	0.1	1.7
Regenerating conifer	13,740.6	473.2	1.2	111.4	175.7	75.6	35.8	208.9
Recently burned	101.5					0.0	0.0	0.0
Shrub	2011.6	85.7	40.7	273.7	71.2	72.0	57.3	137.7
Herbaceous	219.5	6.1	0.2	40.9	273.0	0.0	0.2	38.9
Wetland	112.2	3.7		42.4	1.5	264.9	50.6	4.3
Water	184.7	2.5		48.1	4.7	43.3	941.0	56.8
Barren ground	508.6	8.8	8.4	66.0	21.3	7.9	53.2	290.2

Table 3.4 Area (ha) and percent of classes in the 1915 landcover map that would be impacted by MPB (using all mortality bins) a century later.

MPB severity	Mature conifer	Broadleaf/ mixed wood	Regenerating conifer	Recently burned	Shrub	Herbaceous	Wetland	Water	Barren ground	Grand Total
1–10%	163.2	0.1	593.7	7.7	26.1	1.1		0.1	9.1	801.1
11–20%	256.7	3.5	1042	7.6	113.9	5.5	0.4	2.8	38.7	1471.1
21–30%	229.5	4.8	1301.3	13.1	102.5	20	1.3	0.8	50.1	1723.3
31–40%	212.8	6.9	1207.4	8.9	131.2	12.6	0.5	0.3	35.7	1616.4
41–50%	159.3	10.6	1479.9	18.4	93	3.3	0.5	1.7	37.3	1804
51–60%	217.6	6.1	1225.8	11.6	51.2	27	0.3	0.6	29.9	1570.3
61–70%	168.8	0.2	839.5	0.6	24.4	0.5	0.4	0.8	19.4	1054.5
71–80%	127.5	2.6	648	1.6	16.7	17.3	0.3	0.8	23.8	838.6
81–90%	68.6	5.2	325.7	0.7	30.3	33.3	0	0.3	4.6	468.5
91–100%	146.8	0.1	273.3	0.1	101.8	11.9	2	2.4	22.1	560.5
Total area	**1751**	**40.1**	**8936.5**	**70.2**	**691.1**	**132.4**	**5.6**	**10.6**	**270.7**	**11908.4**
Percent area	**32.90%**	**33.70%**	**60.30%**	**69.20%**	**25.10%**	**22.90%**	**1.20%**	**0.80%**	**28.10%**	**45.10%**



Landcover class		Area change (ha)	Percent change
Recently burned		101.5	99.9
Regenerating conifer		13740.6	92.7
Shrub		2011.6	73.2
Broadleaf/mixedwood		80.0	67.1
Barren ground		508.6	52.7
Herbaceous		219.5	37.9
Wetland		112.2	23.4
Water		184.7	14.4
		16958.7	

Figure 3.5 Map showing area of 1915 landcover classes that transitioned to mature conifer in 2020/22. Blank areas on the map did not transition to another class.

The MPB-adjusted 2020/22 landcover map shows that MPB-affected conifer (7630.5 ha) comprises 28.9% of the study area (Table 3.2; Fig. 3.4). These changes occur at the expense of mature conifer, which decreases from 21,854.7 ha (82.7%) to 14,209.1 (53.8%). Mature conifer had the highest area-weighted mean shape index (23.69) and the largest area-weighted mean patch size (6381.8 ha).

Discussion

The results of our analysis show that the extent of MPB-impacted lodgepole stands (using the 31–100% mortality bins) was approximately half the area of regenerating conifer in 1915 that converted to mature conifer following the fires of 1889 and 1905 (Figs. 3.3 and 3.5). It is likely, then, that the MPB will have significant influence on future ecosystem composition in the study area. The outbreak has also reduced the area-weighted mean patch size for mature conifer from 15,228 ha in 2020/22 landcover map to 6381.8 ha in the MPB-adjusted 2020/22 landcover map. Further, the area-weighted mean shape index of 23.69 for mature conifer in the MPB-adjusted 2020/22 map is similar to the area-weighted mean shape index of 25.91 for regenerating conifer in the 1915 landcover map. The novel MPB disturbance has therefore returned some heterogeneity to the landscape by reducing the area-weighted mean patch size of the mature conifer and reintroducing a comparable amount of spatial complexity to the dominant landcover class (Table 3.2, Fig. 3.4).

The legacy of the frequent, mixed-severity fire regime in Jasper NP is evident on the 1915 landcover map, with over half over the study area comprising regenerating conifer (56.4%). Prior to the policy of fire suppression established circa 1913, large stand-replacing fires occurred on average every 65 years since the Little Ice Age in the study area (Chavardès and Daniels 2016; Tande 1979). The most recent of these large fires occurred in 1889 and 1905, resulting in older cohorts of regenerating conifer (i.e., 26 years old) visible in the 1915 imagery adjacent to the Miette River from the town center to the western park entrance, and younger cohorts (i.e., 10 years old) east of the town and west of the Athabasca River (Chavardès and Daniels 2016; Tande 1979). Large areas of shrub (10.4%) on the eastern side of the Athabasca River and north of its confluence with the Snaring River contribute to the open, patchy configuration of the landscape in 1915. Stands of mature conifer, with an area-weighted mean patch size of 226.5 ha, comprise 20.1% of the study area.

Continuous coniferous species cover dominates the study area in the 2020/22 landcover map (82.7%), with fewer and larger patches as a consequence of the altered fire regime since 1913. In the absence of frequent fire, 92.7% of regenerating conifer in the 1915 landcover map (i.e., over half the study area) succeeded to mature conifer forest. Similarly, a large percentage of shrub (73.2%) also transitioned to the mature conifer class. Smaller areas of transition to mature conifer occurred across all the other classes. However, some of these changes are a result of obscuring issues associated with georeferencing oblique images, especially for the barren ground and water classes, which affected the accuracy of the 2020/22 landcover map. For example, although portions of barren ground, which includes exposed soil, would have likely transitioned to mature conifer, other areas that include rock that were previously visible in the historical photographs are now obscured by mature conifer. This issue occurs frequently north of the Miette River and includes areas where the full extent of lakes have also become obscured by mature conifer.

The spatial extent of the 1889 and 1905 fires is evident in the pattern of regenerating conifer in the 1915 landcover map (Fig. 3.4). Our results show where regenerating conifer (and other landcover classes) in the 1915 landcover map successfully transitioned to mature conifer forest in the 2020/22 landcover map (Fig. 3.5). Given the dates of the two stand-replacing fires, the age class for this mature forest would have been between 105 and 121 years old when the MPB outbreak started in the park circa 2010. Stands of this age class are considered highly susceptible to MPB in epidemic phase (Safranyik and Carroll 2006; Taylor and Carroll 2004).

Within the study area, we estimate that 11,484 hectares of mature conifer were impacted by MPB. This value is smaller than the total area reported by the VRI data because it only accounts for the areas of mature conifer overlapping in the 2020/22 landcover map. This discrepancy relates to some of the VRI polygons comprising multiple tree species (i.e., lodgepole pine may be the dominant species within the polygon, but other species such as white spruce or Douglas fir may also be present). Stands with a high percent of mortality (i.e., > 70%) occur throughout the study area (see Fig. 3.3), specifically south and east of the townsite, the middle portion along both sides of the Miette River, north of the Miette near the western entrance, and the upper elevations toward the northern extent. In stands of high mortality, field surveys undertaken by the University of Lethbridge between 2021 and 2023 indicate a lack of conifer seedling regeneration (Tristan Skretting, personal communication, 26 April 2024). These findings are consistent with previous studies investigating seedling recruitment in lodgepole forests with high MPB mortality in central BC and western Alberta (Astrup et al. 2008; Lieffers et al. 2023; McIntosh and Macdonald 2013).

Time will tell how the forests in Jasper NP will recover from this recent novel disturbance and what the future species composition will look like. A key difference between the processes of MPB and fire is that the latter only disturbs pine species (Meyer et al. 2021). However, a study by Dordel et al. (2008) in Banff and Kootenay National Parks suggests that MPB infestations can have similar effects on stand structure to those of fire, and lead to more open and diverse forests (also see McCullough et al. 1998). In Jasper NP, lodgepole pine stands with low MPB mortality could slowly diversify as canopy openings result in releases of more shade-tolerant species such as Douglas fir and white spruce from the understory (Canham 1989). Stands with higher mortality could see further diversification due to the limited seedling recruitment of lodgepole pine or conversion to herbaceous ecotypes (Lieffers et al. 2023). Monitoring programs can track these changes and be used to inform management.

New compounding disturbances will also influence the recovery and future composition of species in the study area. The timing and intensity of these disturbances can affect the material legacies required for recovery after the initial disturbance (Paine et al. 1998). Since this analysis was completed, the 33,108 ha Jasper Wildfire Complex burned through the upper Athabasca River valley in late July 2024, including 3392 ha (12.84%) of the study area. Approximately 70% of the total area burned by the Jasper Wildfire Complex was in forest stands affected by MPB (Government of British Columbia 2022; Jasper National Park 2024). Although the fire tragically destroyed up to 30% of physical infrastructure in the town of Jasper, it may benefit the recovery of lodgepole stands impacted by MPB. Lodgepole pine are a fire-adapted species and are reliant on fire to release cohorts of unopened cones on their branches and expose viable seedbeds by removing prohibitive understory, forest litter, and feathermosses (Astrup et al. 2008; Lieffers et al. 2023).

However, without fire, lodgepole pine seeds typically become unviable 20 years after the trees die. This scenario could increase the likelihood of species shifts in stands with higher MPB mortality. In contrast, fir and spruce species can only produce viable seedlings from living trees. Hence, a high severity fire that occurs in high mortality MPB stands after lodgepole seedlings become unviable, and kills fir and spruce species, could initiate a transition to non-forest in some areas (i.e., Parks et al. 2019).

Conclusion

Disturbance is a vital attribute of spatial and temporal variability in most ecosystems (Landres et al. 1999). The dominant disturbance in the study area since the Little Ice Age has been fire, which has strongly influenced the HRV of the montane ecoregion in the park. The landscape adapted to this disturbance, and Fig. 3.4 demonstrates the successful regeneration of coniferous species following the large 1889 and 1905 fires. The implementation of fire suppression policies over the previous century resulted in ecosystems departing from their HRV as fire frequency decreased considerably, leading to increased homogeneity on the landscape characterized by mature, closed canopy coniferous forests (Rhemtulla et al. 2002). This departure from HRV, because of the altered fire regime, has directly contributed to the considerable extent of the novel insect disturbance. With an intact fire regime, reflecting landcover composition and patterning evident in 1915, the extent of the MPB in the study area would likely be significantly different.

This study indicates that the novel insect disturbance in Jasper NP can restore landcover composition and pattern lost to fire suppression in the montane ecoregion. However, successional responses in stands with high MPB mortality and the timing of future disturbance could lead to species shifts or transitions to non-forest. Coupled with climate change–driven conditions, these potential transitions could increase the departure from the HRV and shift the composition, function, and pattern of landscapes under Parks Canada's management. It seems plausible, perhaps likely, that montane ecosystem in Jasper will undergo significant transitions that will necessarily involve shifts in how they are managed. Increased novelty suggests managing with history in mind but not necessarily as a specific target (Higgs et al. 2014).

Acknowledgements
We thank Spencer Rose, Mary Sanseverino, Mike Whitney, and Claire Wright for their important work in developing the tools used in this analysis. We also thank Rachel Dietrich, Mike Eder, Katelyn Fryer, Kristyn Lang, Catalina Madrid, Shima Tajarlou, and Sonia Voicescu for their assistance in collecting field data. We thank Jasper National Park, fRI Research, Mitacs Accelerate, Lorene Kennedy Environmental Studies PhD Award, Lorene Kennedy Field Research Award, and an SSHRC Insight research grant (435-2016-0667) for supporting this research.

Chapter 4: Beyond history: Anticipating the influence of a novel disturbance regime on future landcover change in Jasper National Park.

Abstract

Disturbances and recovery processes are closely linked phenomena, and changes that alter this linkage can result in unexpected outcomes. Across North America, climate change and management practices are driving changes in the frequency, severity and extent of forest disturbances in protected areas. In Jasper National Park, two recent disturbance events involving novel parameters and agents have impacted large swaths of the montane ecoregion (i.e., lower elevations where the climate is typically warmer and drier). There is increasing concern about how ecosystems will respond to novel disturbances. Novel responses that involve changes in species assemblages or regime shifts to non-forest could degrade ecological integrity in the park and alter important ecosystem services. Traditional approaches (i.e., adaptive management, historical range of variability) to manage these changes may prove insufficient. To address uncertain trajectories, we undertook a scenario planning exercise to conceive plausible futures in Jasper National Park. We developed four scenarios based on different levels and combinations of ecological novelty and management intervention and considered their implications to enable managers to compare outcomes for different combinations of alternative management strategies and responses to novel disturbance. The purpose of this planning exercise is to better understand how these two variables could influence the future of montane ecosystems in Jasper NP and by extension find value for scientists and managers who are contending with increasing novelty in wildlands and protected areas.

Introduction

The 2024 Jasper Wildfire Complex in Jasper National Park (hereafter "Jasper NP"), Alberta highlights an emerging trend of fast-moving, high-severity wildfires occurring in many parts of the world (Fig 4.1; Balch & Williams 2024). These fires typically ignite during extreme weather conditions—hot, dry, and windy—and are characterized by their rapid growth and high intensity, which often overwhelm firefighting efforts (Jolly et al. 2015; Tedim et al. 2018). The Jasper Wildfire Complex grew to 30,000 hectares (ha) within 48 hours, by which time it reached the town of Jasper and destroyed hundreds of structures. This fire exhibited similar behaviour to several other destructive wildfires that have burned in Alberta over the past decade, including the 2011 Slave Lake, 2016 Fort McMurray, and 2017 Kenow wildfires. Extreme fire events are also occurring in other regions within Canada, the US, Australia and Europe, threatening human communities and causing significant ecological consequences (Balch et al. 2024; Collins et al. 2021; García-Llamas et al. 2019).

Figure 4.1 Looking north from Wabasso Road toward Pyramid Mountain on August 19, 2024 after the Jasper Wildfire Complex. Photo credit: Jasper National Park.

Climate change is an important driver of global shifts in fire regimes that lead to extreme fire behaviour. Recent studies have observed climate-driven increases in fire weather season length, extreme fire weather frequency, lightening ignitions, fire spread rates and fire severity (Coop et al. 2022; Jones et al. 2022; Parks & Abatzoglou 2020; Pérez-Invernón et al. 2023; Wasserman & Mueller 2023; Whitman et al. 2022). Consequently, many regions have experienced an increase in large fires and total burned area (Abatzoglou & Williams 2016; Bedia et al. 2015; Krawchuk et al. 2009). However, fire exclusion and suppression practices have also influenced fire behaviour by altering forest structure and fuel availability (Hagmann et al. 2021). In the absence of low- and moderate-severity fires, both live and dead fuels have increased on the landscape, contributing to higher severity fires (e.g., the Jasper Wildfire Complex; Parks et al. 2018; Parks et al. 2025).

These drivers of change are also altering the extent and severity of other forest disturbances (Cobb & Metz 2017; Dale et al. 2000; Seidl et al. 2017). In the late 1990's, the largest recorded outbreak of mountain pine beetle (MPB; *Dendroctonus ponderosae*), a forest insect native to western North America, erupted in central British Columbia (Axelson et al. 2009). Changing climatic conditions and an abundance of mature lodgepole pine (*Pinus contorta;* the main host species of MPB), due to decades of fire suppression, facilitated unprecedented range expansion eastward and northward (Carroll et al. 2006; Cullingham et al. 2011; Kurz et al. 2008; Taylor & Carroll 2003). In western and northern Alberta, the widespread impacts of MPB may have been exacerbated because naïve hosts have not evolved specific defense responses (Burke et al. 2017; Cudmore et al. 2010; Raffa et al. 2008). In Jasper NP, which had no prior records of MPB, up to 70% of lodgepole pine in the park were affected by the outbreak (Dalman 2004; Skretting 2024).

The altered fire regime and unprecedented MPB outbreak in Jasper NP represent a significant departure from the disturbance dynamics that have occurred historically (Tricker et al. 2025). Ecosystems have constantly changed in response to these dynamics, but bounds of natural variability emerge over time as species adapt to disturbance regimes (Johnstone et al. 2016; Morgan et al. 1994; Turner & Seidl 2023). The concept "historical range of variability" (HRV) was developed to improve understanding of ecosystems and their long-term variation prior to anthropogenic change (Landres et al. 1999). HRV can provide natural resource managers with historical references from which to assess changes in contemporary conditions and inform restoration goals for degraded ecosystems (Keane et al. 2009; Landres et al. 1999).

In recent decades, the value of historical references and HRV has been called into question for managing (or restoring) environments undergoing rapid change (Harris et al. 2006; Higgs et al. 2014; Millar et al. 2007; Stephenson et al. 2010). This issue is pertinent to Jasper NP because the emergence of novelty in disturbance regimes raises concerns about how ecosystems will respond to these changes (Johnstone et al. 2016; Newman 2019; Turner 2010; Turner & Seidl 2023). Disturbances and recovery processes are closely linked, so changes that alter this linkage could result in novel outcomes (Turner 2010; Seidl & Turner 2022). Additionally, interactions between two novel disturbances (and other drivers of change such as climate change or invasive species) may increase the likelihood of novel outcomes (Turner & Seidl 2023). However, the potential for fundamental ecological change (i.e., regime shifts or novel ecosystems) is greatest when novelty occurs in both the disturbance and the ecological response to the disturbance (Seidl et al. 2016; Turner & Seidl 2023).

The Canada National Parks Act (2000) stipulates that the first priority of park management is to maintain "ecological integrity" – a concept underpinned by HRV and defined as "a condition that is determined to be characteristic of its natural region and likely to persist, including abiotic components and the composition and abundance of native species and biological communities, rates of change and supporting processes". To fulfill this mandate, the ecological integrity monitoring program provides a framework for identifying measures that track key ecosystem components and processes in the park (Parks Canada 2024a). Restoration guidelines can then be used by managers to define goals for measures that are in decline or poor condition and select appropriate interventions through detailed planning (Parks Canada 2008).

However, this approach fails to consider the potential for novel disturbances and responses to create extraordinary surprises, and traditional management approaches to maintain or restore ecological integrity may prove ineffective. Therefore, different approaches are required for managing the uncertainty associated with increasing ecological novelty (both successional responses and future novel disturbances) in Jasper NP. In this paper, we undertake a scenario planning exercise to consider plausible futures in the montane ecoregion. We developed four scenarios that describe likely ecological outcomes based on different combinations of analog and novel responses (to recent and future disturbances) and differing levels of management intervention. The purpose of this exercise is to better understand how these two variables could influence the future of montane ecosystems in Jasper NP and by extension find value for scientists and managers who are contending with increasing novelty in wildlands and protected areas.

Site description

Jasper NP was established in 1907 and extends over 11,000 km^2 of the eastern Canadian Rockies in west-central Alberta. Historically, the montane ecoregion experienced frequent fire return intervals comprising mixed-severity fires (Chavardes & Daniels 2016; Tande 1979). However, a considerable decline in fire return interval has occurred in the park since fire suppression began in 1913 (Murphy et al. 2007). Quantitative analyses of historical (captured in 1915) and repeated contemporary photographs indicate that landcover has homogenized in the montane ecoregion (i.e., lower elevations) in the absence of fire (Tricker et al. 2025). Grasslands and shrub decreased in extent and juvenile and open forest stands transitioned to dense, closed-canopy stands comprising mature, even-aged coniferous species (Rhemtulla et al. 2002). This shift to late-successional species can be attributed to the departure from the historical fire regime in Jasper NP, which is consistent with changes observed in fire return intervals across the Canadian Rockies (Chavardes & Daniels 2016; Rogeau et al. 2016; Tricker et al. 2025).

A consequence of these profound landcover changes in Jasper NP is increased susceptibility to high severity fire and insect outbreaks (Parks et al. 2018; Rhemtulla et al. 2002; Taylor & Carroll 2003). During the range expansion of MPB into west central Alberta during the late 1990's, the insect was first observed in Jasper NP in 1999 near the northwestern boundary (Dalman 2004; de la Giroday et al. 2012; Sambaraju & Goodsman 2021). Towards the end of the following decade, the MPB population reached epidemic levels throughout the montane ecoregion, impacting approximately 229,000 ha of lodgepole pine, before a sharp population decline occurred in 2018 (Skretting 2024). Since the end of the outbreak, there have been two large, high-intensity wildfires in Jasper NP. The 2022 Chetamon wildfire occurred 15 km north of the town of Jasper, consuming 6,456 ha of montane and subalpine forest. Two years later, multiple lightning strikes ignited the 2024 Jasper Wildfire Complex, which went on to burn 32,722 ha and tragically consumed 30% of infrastructure in the Jasper townsite and numerous outlying facilities (Parks Canada 2024b). Both the Chetamon wildfire and the Jasper Wildfire Complex burned through stands of MPB-affected lodgepole pine. These fuels likely contributed to fire behavior, but wildfire severity was primarily driven by extreme weather conditions and low fuel moisture content (Parks Canada 2025; Simard et al. 2011).

Methods

Scenario planning is a systematic method that considers a variety of possible outcomes based on the uncertainties in the system (Peterson et al. 2003). Each scenario describes an alternative, dynamic story of a possible future that captures key elements of the system uncertainties (Chapin et al. 2010; Peterson et al. 2003). This approach is designed to provide insight into drivers of change, uncover the implications of current trajectories, and help identify options for management actions (Higgs et al. 2014; Peterson et al. 2003, Turner 2010). In this section, we follow elements of Petersen et al's (2003) scenario development process to identify our focal issue and assess the system to determine which uncertainties are most likely to affect the focal issue. We then identified alternative ways that the system could evolve to develop each scenario, which are described in the results section.

Identification of the focal issue

The focal issue for this planning exercise is the threat of recent (and future) novel disturbance in Jasper NP to drive novel outcomes that could compromise the ecological integrity approach followed by Parks Canada. In the montane ecoregion, the potential for regime shifts and the emergence of novel ecosystems in response to novel disturbance is a pressing concern. Regime shifts occur when an increase in a continuous driver of change in an ecosystem exceeds a tipping point (Hallett et al. 2013). This causes an abrupt shift to an alternative state, where the core ecological functions, structures and processes of the ecosystem are fundamentally changed (Andersen et al. 2008). Similarly, novel ecosystems are defined as self-organizing ecosystems that differ significantly (i.e., species composition or ecological functioning) from their historical state due to anthropogenic activities (Hobbs et al. 2013). However, a key difference between regime shifts and novel ecosystems is the irreversibility of the latter due to the practical limitations of recovering historical qualities.

Regime shifts and novel ecosystems have consequences for maintaining ecological integrity (e.g., rare species may decline further due to reduced habitat and increased predator interactions) and the provision of ecosystem services (e.g., diminished carbon sequestration, loss of scenic values).

Further, future novel disturbance events, such as large, high-intensity wildfire, can compound ecological changes and pose a significant risk to residents and visitors in the park (Paine et al. 1998). Forced closures of the park due to wildfire would have a large impact on the tourism-driven local economy (there were 2.48 million visitors to the park in 2023; Parks Canada 2023a). These closures would also disrupt interprovincial travel corridors and impact the flow of goods (approximately 1.8 million vehicles and 16,000 trains travel through Jasper NP per year; Government of Alberta 2025; Railstate 2024).

Assessing the system

Our system assessment is focused on identifying uncertainties associated with the two key variables which are most likely to impact our focal issue: ecological novelty and management intervention. The first part of the assessment evaluates the two dimensions of ecological novelty from Turner and Seidl (2023): novel disturbance agents and novel disturbance parameters (Turner & Seidl 2023). The first dimension concerns novel disturbance agents, both abiotic and biotic, which were not present historically (Turner & Seidl 2023). Because ecosystems haven't coevolved with these disturbances, they may lack the ability to resist or recover from them. The second dimension relates to attributes of disturbances that fall outside their HRV, such as changes to disturbance return interval, duration or intensity/severity (Turner & Seidl 2023). This determination requires historical data to quantify attributes of disturbance regimes along with the configuration and composition of ecosystems over space and time (Keane et al. 2009).

To depict the effects of novel disturbance on landcover in Jasper NP, we focused on a study area centered on the town of Jasper in the montane ecoregion. We adapted historical and contemporary landcover maps from Tricker et al. (2025), which were originally produced using a recently developed workflow for classifying and georeferencing oblique images (Tricker et al. 2024). We clipped these data to a new extent to expand landcover coverage adjacent to the townsite and included the extent of the 2024 Jasper Wildfire Complex (Figure 4.2).

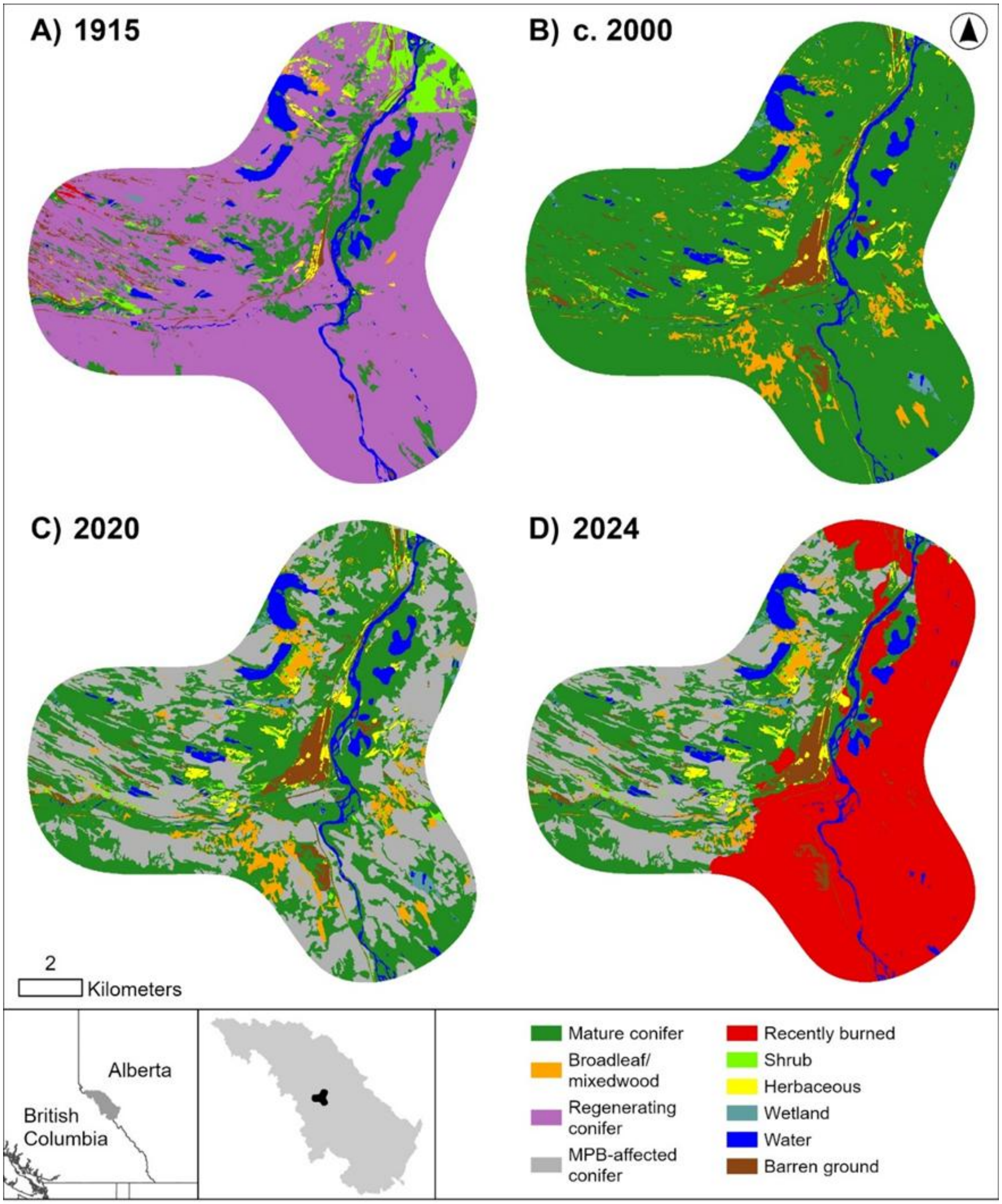


Figure 4.2 Landcover change and recent disturbance in the montane ecoregion of Jasper National Park. Maps are a composite of classified and georeferenced historical and repeated images adapted with permission from Tricker et al. 2025. The historical images were captured by the Dominion Land Survey in 1915 and repeated by the Mountain Legacy Project (https://mountainlegacy.ca/) in 2020/22. The workflow for developing these maps first appeared in Tricker et al. 2024. Map A shows landcover in 1915 is recovering from stand replacing fires in 1889 and 1906. Map B shows that contemporary landcover has transitioned to mature conifer. Map C shows the extent of the MPB impacts that occurred over the previous decade. Map D shows the extent of the 2024 Jasper Wildfire Complex.

The MPB represents a novel disturbance agent in Jasper NP, as there are no records of the insect in the park prior to 1999 (Dalman 2004). In lodgepole pine forests with no historical precedent for MPB, naïve host trees may lack the defence responses of trees that have coevolved with MPB in their native range (Raffa et al. 2008; Burke et al. 2017). Robinson (2015) found that MPB impacts in novel habitats were 1.7 – 3.9 times greater than those that had coevolved with the insect. As a result, the elevated susceptibility of naïve hosts in Jasper NP may have contributed to the considerable MPB impacts in the montane ecoregion (Fig 4.3). However, tree mortality varies throughout the extent of affected forest, which can replicate some of the effects of mixed-severity fire and lead to different successional responses across the landscape (McCullough et al. 1998; Tricker et al. 2025).

In forest stands that incurred high MPB mortality in the montane ecoregion, recent field surveys have observed poor lodgepole seedling recruitment (Skretting 2024; Skretting et al. 2025). These findings are consistent with previous studies investigating seedling recruitment in lodgepole forests with high MPB mortality in central BC and western Alberta (Axelson et al. 2018; Astrup et al. 2008; Lieffers et al. 2024; McIntosh & Macdonald 2013). Therefore, a transition from pine to broadleaf species such as aspen (*Populus tremuloides*) or birch (*Betula*), and other more shade tolerant coniferous species such as white spruce (*Picea glauca*) is likely (Axelson et al. 2018; Bassil et al. 2024; Lieffers et al. 2024).

The 2022 Chetamon wildfire and 2024 Jasper Wildfire Complex in Jasper NP both have novel disturbance parameters (Fig 4.3). Prior to the start of fire suppression in 1913, low-to medium-intensity fires were widespread and frequent in the montane ecoregion, with large, high-intensity fires occurring every 65.5 years (Tande 1979). Since 1913, the considerable decline in fire periodicity has resulted in increased homogeneity on the landscape (Fig 4.3; Rhemtulla et al. 2002; Tricker et al. 2025). The build-up of continuous fuels, coupled with increased frequency and duration of extreme fire weather, contributed to the high intensity of these two fires. High severity fires may elicit uncertain ecological responses, especially where wildfires interact with beetle-killed forests (Buma et al. 2013; Coop et al. 2020). A concern with compounding disturbances (i.e., two disturbances occurring in short period of time) is the potential for unpredictable outcomes that lead to regime shifts (Buma 2015; Paine et al. 1998; Simard et al. 2011).

Within the Jasper Wildfire Complex perimeter, cones opened by the fire could facilitate regeneration of dense pine stands. However, seedling recruitment may be limited due to the severity of the fire in certain areas (see Busby et al. 2020). Further, the large patch size of this fire may reduce the effectiveness of coniferous seed delivery from adjacent forests (whereas aspen and birch can regenerate by sprouting from stumps and the roots of burned trees; Lindenmayer et al. 2011). Future drought stress may also have an impact on recruitment success (Harvey et al. 2016; Hoecker & Turner 2022; Whitman et al. 2019). Hence, a transition to other tree species (both deciduous and coniferous) or non-forest vegetation (e.g., shrub or grassland) is also possible within the fire perimeter (Coop et al. 2020; Stevens-Rumann et al. 2018; Stralberg et al. 2018).

The second part of the system assessment focuses on approaches for managing increasing ecological novelty in Jasper NP. Although Parks Canada policy states that national park ecosystems are to be managed with minimal interference to natural processes, active management may be allowed to restore ecological integrity when the structure and function of ecosystems have been severely altered (Parks Canada 2023b). Additionally, naturally occurring processes (e.g., fire, insects, and disease) may be manipulated when no reasonable alternative exists to prevent adverse effects to adjacent lands, reduce threats to park facilities and public safety, and meet the management objectives for natural features (Parks Canada 2023b). Therefore, policy can support decisions by managers in Jasper NP to intervene and actively manage novel disturbance in the park and limit novel ecological responses. However, any such decision requires careful assessment of ecological novelty in relation to the concept of ecological integrity, and other practical intervention considerations such as the availability of funding and resources, public support, etc.

In recent decades, Parks Canada has proactively managed the threat of destructive disturbances. Both control (i.e., prescribed burning, pheromone baiting, and the cutting and burning/removal of colonized trees) and non-control of MPB is considered compatible with the ecological integrity mandate (McFarlane et al. 2004). While there is no historical precedent of MPB in Jasper NP, the insect is native to other areas of the southern Canadian Rockies and plays an important role in rejuvenating forest ecosystems. However, the recent MPB outbreak is also symptomatic of an unhealthy ecosystem because fire suppression has created even-aged, mature forests (Trzcinski & Reid 2008). Jasper NP undertook direct control measures to slow or limit the spread of MPB through the park (and into adjacent lands; Jasper National Park 2016).

Jasper NP has also implemented wildfire risk reduction measures in recent decades to protect the town of Jasper, visitor facilities, outlying commercial operations and other critical infrastructure in the park. FireSmart initiatives and fuel management projects adjacent to the townsite have employed mechanical thinning and dead vegetation removal to treat 1,700 ha of forest since 2003. These actions are intended to reduce crown fire potential, fire intensity and rates of spread. The park has also established and maintained various fireguards in the park, which can slow the spread of fire and serve as a line of defense for firefighting. Prescribed fire is another important tool for managing fuel loads, creating natural fire breaks and restoring fire-adapted ecosystems. Since 1994, Parks Canada has conducted 45 prescribed burns to treat 4,600 ha in the park. Finally, the delineation of fire management zones in the park, based on values at risk, potential fire behaviour, and wildfire barriers, dictates specific response strategies and tactics for wildfires based on their location.

Identification of alternative future trajectories

The effects and interactions of novel disturbance in Jasper National Park raise concerns about what the future holds, with respect to both successional outcomes in recently disturbed areas and the persisting threats of novel disturbance. The Jasper Wildfire Complex burned through much of the upper Athabasca River valley, and in certain areas all organic matter down to mineral soil was consumed (C. Watson, personal communication, October 29, 2024). This landscape could take decades to recover and may see new species assemblages emerge. Additionally, in unburned areas of the montane ecoregion, successional trends in the beetle killed forests may see species shifts depending on the amount of mortality in affected stands (Skretting et al. 2025). These unburned areas also constitute a continuing fire risk to the town of Jasper and park infrastructure.

To identify alternative ways the system could develop, we considered different combinations of ecological responses (analog and novel) to recent and future disturbance and management approaches (inaction or intervention). To illustrate these combinations, we plotted these variables on axes in Fig 4.3. These scenarios are intended to promote discussion of options and not necessarily to reflect likely future conditions. To this end they represent fixed positions in a state-space that varies in intensity and within each scenario there can be variability along either axis.

Building and testing scenarios

We then developed our scenarios using insights gained from the assessment process to flesh out the alternative ways the system could evolve. To ensure plausibility, we focused on continuity between historical and present events with hypothetical future events. We then refined each scenario by testing the consistency of our narratives through the expert opinion of the author team (Peterson et al. 2003).

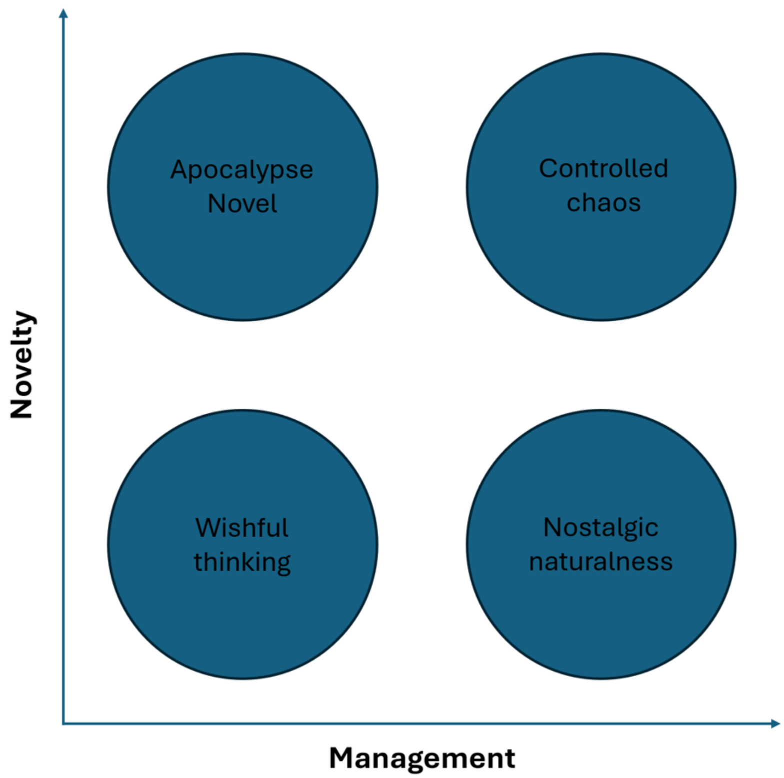


Figure 4.3 Conceptual diagram that plots four scenarios in relation to degrees of novelty and management intervention. The Y-axis represents increasing ecological novelty and the X-axis represents increasing management intervention.

Results

1. Wishful thinking

Description: Montane ecosystems successfully recover from recent disturbance events and the landscape returns to 20th century homogenous forest cover dominated by lodgepole pine. Management actions under this scenario would be limited and focus primarily on suppressing fires in high priority areas of the park.

Implications: Low management input is required, and outcomes would meet visitor expectations as homogenous forest cover has defined visitor experiences for multiple generations. However, increasing homogeneity in the montane ecoregion will decrease biodiversity and degrade adaptive capacity for future disturbance events (Hewitt et al. 2009; Viljur et al. 2022). Additionally, continuing to suppress fire in the montane ecoregion would degrade ecological integrity because Parks Canada acknowledges that wildfire is an important natural process (Parks Canada 1997).

2. Nostalgic naturalness

Description: Intensive management actions are undertaken to return ecosystems to their HRV and increase heterogeneity across the landscape. Integral to this scenario is restoring the effects of the historical fire regime (i.e., frequent, mixed severity fire) through the extensive use of commercial logging, mechanical thinning, brush clearing and prescribed fire. Once these actions are completed, managers can allow future wildfires to burn when conditions permit to maintain HRV in the montane ecoregion. However, if specific targets for area burned via wildfire are not met, then further management intervention would be required.

Implications: Reintroducing fire (and its effects) to restore ecosystems to their HRV would improve ecological integrity in the park because fire is considered an important natural process. Wildfires create heterogeneity on the landscape, which can increase biodiversity through the provision of more diverse resources and habitats and reduce the severity and impacts of future disturbances (He et al. 2019; Turner 2010; Turner et al. 2013). This scenario could also support reconciliation with Indigenous Peoples connected to the park by reintroducing indigenous burning practices that were integral to the historic maintenance of HRV (Dickson-Hoyle et al. 2021; Nowacki et al. 2012). However, certain management actions (e.g., commercial logging and mechanical thinning) undertaken to restore HRV would be very expensive and may be viewed as antithetical to the wilderness qualities of Jasper NP. Other concerns associated with this scenario include public health issues related to elevated smoke and particulate matter during prescribed fire season and the inherent risks of allowing future wildfires to burn in the montane ecoregion.

3. Apocalypse novel

Description: Unbridled ecological novelty is the hallmark of this scenario. Ecological responses to the recent MPB outbreak and wildfires result in fundamental change in the dominant cover in the montane ecoregion, characterized by shifts to new species assemblages or alternative stable states (i.e., non-forest). Climate-driven novel disturbances (e.g., drought, floods, wildfire, insect outbreaks; non-native invasive species) occur more frequently under this scenario, and trigger further ecological transformations by a) acting as destabilising (reinforcing) feedback or b) through the compounding effects of short interval disturbances (Coop et al. 2020; Paine et al. 1998; Parks et al. 2025; Raffa et al. 2008; Turner & Seidl 2023; Whitman et al. 2019). Novel disturbances such as wildfire and floods also constitute a significant risk to visitors and residents, resulting in frequent park closures and damage to park infrastructure, residential buildings, commercial businesses, and national transport routes. The relative lack of management action under this scenario is a consequence of falling revenues in the park and public resistance to broad-scale ecological interventions.

Implications: There is considerable uncertainty concerning the implications of unfettered ecological novelty in Jasper NP. The emergence of alternative trajectories in the montane ecoregion could affect ecological integrity in numerous ways, such as significantly altering the trophic structure of the montane ecoregion through changes to habitat and the availability of resources. For example, a decline in forest cover could negatively impact woodland caribou through loss of habitat and increased exposure to predation (Nagy-Reis et al. 2021). However, a greater diversity of habitats and resources could potentially increase biodiversity in the montane ecoregion (Evers et al. 2018; He et al. 2019). The provision of ecosystem services could also be significantly affected under this scenario. For example, regimes shifts (i.e., conversions to non-forest) could negatively affect aesthetic values for park visitors, increase surface runoff that degrades water quality, and decrease carbon sequestration (Cooper et al. 2020; Standish et al. 2013). However, non-forested ecosystems could also mitigate the impacts of future novel disturbance such as wildfire, windthrow or insect outbreaks. Increases in park-wide closures or damage to critical infrastructure due to novel disturbance events will disrupt park visitation and interprovincial freight travel, resulting in substantial local and national economic impacts. The costs of repairing park infrastructure, commercial operations and residential buildings damaged by novel disturbance events such as wildfire and floods are also likely to be significant. Additionally, Jasper NP may be liable for novel disturbance events that originate in the park and spread to adjacent lands.

4. Controlled chaos

Description: As ecological novelty increases in the montane ecoregion, aggressive management intervention is used to maintain important ecosystem functions and services and protect resources from future novel disturbance. Managers use prescribed fire and tree planting, especially drought and fire-resistant species (i.e., *Picea glauca*, *Populus* varieties and genetically modified species), to aid the recovery of areas affected by the recent wildfires and MPB outbreak and maintain habitat for at-risk species such as woodland caribou. However, to reduce fire risk closer the town of Jasper, and adjacent to critical infrastructure, frequent prescribed and cultural burns are used to shift thousands of hectares of recently disturbed areas to non-forest. To reduce fire risk in other areas of the montane ecoregion, the Parks establishes a network of extensive fire roads and large fire breaks (i.e., >10,000 ha) across the landscape. Fire breaks are created using mechanical thinning and prescribed fire during appropriate weather windows. Parks Canada also significantly increases the capacity of firefighting units in the park by hiring additional personnel and purchasing/leasing new equipment including bulldozers, tankers, and fixed and rotor wing aircraft. These aircraft are also used for forest monitoring purposes to rapidly identify the presence of forest pests, and targeted management actions can be employed to prevent outbreaks reoccurring in the park (i.e., prescribed fires, pheromone baiting and cutting and removal/burning of infected trees). A variety of increasingly sophisticated surveillance and rapid response technologies are also introduced, such as automated watchtowers or drones equipped with heat sensors and smoke detectors, which increases costs potentially at the expense of visitor services and ecosystem management programs.

Implications: The inevitable increase in ecological novelty in the montane ecoregion require managers to undertake intervention and adaptation strategies to maintain ecological integrity in the park. While inaction under Scenario 3 results in alternative stable states with different ecological functions, under this scenario managers use interventions to maintain important ecosystem processes and services (i.e., hybrid ecosystems; Hobbs et al. 2014). For example, planting fire-resistant tree species such as Aspen in recently disturbed areas will provide critical habitat for at-risk species and reduce the risk of high intensity fire (Buma & Wessman 2012; White & Zipperer 2010). However, increasing ecological novelty will alter habitats and resources in the montane ecoregion, and result changes to the trophic structure and the extirpation of numerous species. New research can identify species that are most adaptable to increasing ecological novelty and enable managers to better triage resources. Intervention and prevention strategies will require significant increases to operational budgets in the park. Jasper NP could offset these costs will timber sales from mechanical thinning, but additional funding streams will need to be identified to support these strategies.

Discussion

Our four scenarios highlight the myriad challenges and dilemma's facing natural resource managers in Jasper NP (and many other protected areas undergoing rapid change) as they plan for an uncertain future. We considered the implications of each scenario to enable managers to compare outcomes for different combinations of alternative management strategies and responses to novel disturbance. However, there are numerous other considerations associated with these scenarios that could be determinative to the future of montane ecosystems.

Climate change will have a large influence on the likelihood of each scenario. Recent climate projections indicate western Canada will experience rising temperatures, shifts in precipitation patterns, and increased frequency and intensity of extreme weather events (Price et al. 2013). These projections are expected to alter the dynamics of forest recovery and exacerbate future disturbance events, which will lower the likelihood of "Wishful thinking" (Anderson-Teixeira et al. 2013; Li et al. 2000; Seidl et al. 2017; Stevens-Rumann et al. 2018). Plans to restore ecosystems to their HRV and reintroduce wildfire to the montane ecoregion under "Nostalgic naturalness" will also be affected by climate change due to the increased risk of high intensity fires (Whitman et al. 2022; Wotton et al. 2017). Conversely, climate change is a key driver of widespread novel conditions, further increasing the potential for novel responses that lead to new species assemblages or regime shifts described under "Apocalypse Novel" (Kerr et al. 2025; Turner & Seidl 2023). The threat of these significant ecological changes to ecological integrity in the park, and climate driven high-severity disturbance events, may preclude Parks Canada's limited intervention strategy under this scenario. This would then increase the likelihood of "Controlled chaos", whereby Parks Cananda undertake aggressive management action to maintain ecological integrity in the montane ecoregion and proactively address the threat of novel disturbances.

The challenges of restoring ecosystems to their HRV under "Nostalgic naturalness" underscore the changing value of the HRV concept. The significant costs and scale of intervention required to implement and maintain this scenario under a changing climate, coupled with the risks of high intensity fire, validate recent concerns about the continued use of this concept in resource management and ecological restoration (Craig, 2010; Harris et al., 2006; Higgs et al., 2014; Kerr et al. 2025; McGarigal & Romme 2012; Millar et al., 2007). Indeed, Landres et al. (1999) acknowledge concerns about the relevance of HRV to contemporary environments that have fundamentally changed from what they once were. However, Romme et al. (2012) argue that the evolving value of HRV lies in using historical records to understand how ecosystems have responded to environmental variation in the past to guide current and future conservation management. Therefore, managers can use an understanding of ecosystem dynamism provided by HRV to move an ecosystem back within the bounds of various discrete parameters (Higgs et al. 2014).

Multiple processes, including novel disturbances, are driving widespread novel conditions that increase the potential for unprecedented ecosystem trajectories (Kerr et al. 2025; Turner & Seidl 2023). Without intervention, regime shifts to alternative stable states are likely in many areas of the montane ecoregion (i.e., “Apocalypse Novel”). Shifts between ecosystem states are often undesirable because they can alter ecological processes, disrupt ecosystem services, and adversely affect at-risk species (Harris et al. 2013). Further, hysteresis makes regime shifts more difficult to reverse, especially if something external to the system such as climate has changed (Groffman et al. 2006; Scheffer et al. 2001). Irreversibility is a feature of novel ecosystems, and their emergence in Jasper NP could have implications for maintaining ecological integrity in the park (Hallett et al. 2013). Although Parks Canada’s legal guidelines recognize that systems with integrity may exist in different states, changes to ecosystems must occur “within acceptable limits”, which may further preclude Parks Canada’s limited intervention under “Apocalypse Novel” (Wurtzebach & Schultz 2016).

The approach to managing the integrity of ecosystems may be misguided or impractical in an era of increasing novelty and climate change (Keane et al. 2009; Wurtzebach & Schultz 2016). Parks Canada considers ecosystems to have integrity when its native components are intact (i.e., the biotic and abiotic pieces and the frequency and intensity of ecological processes; Parks Canada 2021). Traditional management approaches have often focused on the preservation of specific species, species assemblages, or the condition of habitat required to sustain populations (Safford et al. 2012a). However, many native species may fail to adapt to changing conditions (Visser 2008). Therefore, a more appropriate approach may be better focused on long term adaptive capacity of ecosystems, with emphasis on maintaining structural and functional attributes of ecological integrity instead of the persistence of specific species or their habitat (i.e., “Controlled chaos”; Dudney et al. 2022; Hobbs et al. 2009; Safford et al. 2012a; Wurtzebach & Schultz 2016). Increasing ecological novelty can occur without necessarily producing novel ecosystems. In the case of hybrid ecosystems—comprising novel and historical elements—there may be greater capacity for realizing Parks Canada’s traditional concept of ecological integrity (Hobbs et al. 2013).

Parks Canada policy allows for management intervention to address the adverse effects of ecological processes or restore ecosystems with degraded ecological integrity. Therefore, this policy may support the intervention actions described in “Nostalgic naturalness” and “Controlled chaos”. However, efforts to restore ecosystems to their HRV may lack public support due to unaligned cultural perspectives on naturalness values and the likely impacts to visitor experiences (i.e., park closures due to active fires or health issues related to elevated smoke and particulate matter; Aplet & Cole 2010, Safford et al. 2012a, Wurtzebach & Schultz 2016). Similarly, there are uncertainties related to public attitudes toward increasing levels of intervention to manage emergent ecology novelty (Lemieux et al. 2011). To improve public engagement in decision-making, Parks Canada can introduce new indicators to ecological integrity monitoring that incorporate social and economic considerations (i.e., feasibility, relevance, cost, and acceptance; Tulloch et al. 2011; Wurtzebach & Schultz 2016). This approach would support outcomes from recent studies that indicate important relationships between ecological integrity and human health (Lemieux et al. 2011).

New research can support management objectives that respond to changes rather than resist them. The evolving value of historical ecology (see above) can identify long-term processes that influence ecological outcomes under different climate conditions or disturbance regimes (Safford et al. 2012a). For example, the development of innovative methods used to quantify the composition and configuration of landcover present in historical survey photographs captured in Jasper NP can reveal new information about the mechanisms underlying ecosystem dynamics and resilience (Trant et al. 2020; Tricker et al. 2025). Historical ecology studies can also be used to identify parameters of important ecological processes, determine expected levels of ecosystem services, redefine operational concepts such as ecological integrity or resilience, and suggest appropriate future trajectories (Keane et al. 2009; Landres et al. 1999; Safford et al. 2012a; Safford et al. 2012b; Swetnam et al. 1999). Other research streams can focus on maintaining valued ecosystem services as ecological novelty increases and gauge public tolerance for increased management interventions proposed under “Controlled chaos” (Lemieux at al. 2011).

Conclusion

Scenario planning makes future uncertainty explicit so managers can evaluate the effectiveness of management strategies under a range of possible futures (Peterson et al. 2003). In Jasper NP, the likelihood of increasing ecological novelty in the montane ecoregion could negate traditional management approaches to maintain contemporary conditions or restore historical processes and landcover configurations (Hobbs et al. 2014; Kerr et al. 2025). Thus, “prospective strategies” that focus on anticipated conditions are required (Aplet & McKinley 2017). However, historical knowledge can still play an important role in guiding contemporary decision-making and planning (Higgs et al. 2014; Safford et al. 2012a; 2012b). Concepts such as HRV can be used to understand and quantify how ecosystems have responded to different magnitudes and frequencies of historical disturbance and identify deviations in recent and future ecological responses. Actions can then be undertaken sooner to mitigate the undesired effects of novel disturbance and align management goals with emerging hybrid and novel ecosystems (Turner & Seidl 2023).

Chapter 5: Conclusion

Recurrent disturbances are an important driver of ecosystems dynamics (Turner et al. 2010). Palaeoecological and contemporary evidence indicates that ecosystems are typically resilient to historical disturbances and can recover their characteristic structure and function following perturbation (Holling 1973, Beisner et al. 2003, Johnstone et al. 2016). However, climate change and management practices are driving changes in the disturbance regimes. Because disturbances and recovery processes are closely linked, there is growing concern that novel disturbances may lead to novel outcomes (Turner & Seidl 2023). The research undertaken in this dissertation is focused on understanding the implications of novel disturbance in Jasper National Park (hereafter "Jasper NP"). Chapter 2 describes and tests the accuracy of a new method for deriving georeferenced landcover data from historical and repeated oblique photographs. In Chapter 3, this new method is deployed to quantify changes to the composition and configuration of montane ecosystems. Chapter 4 builds on this analysis by undertaking a scenario planning exercise to consider the uncertainty associated with the successional trajectories that will follow recent novel disturbances.

This concluding chapter begins with summarizing the key findings and contributions of the research undertaken in this dissertation. The second section covers lessons learned from producing these key findings. The third section reports on the methodological limitations of the data and software, followed by a section on what could have been done differently. The final section provides recommendations for workflow improvements, future research and management considerations.

5.1 Key findings

5.1.1 Chapter 2: Assessing the accuracy of georeferenced landcover data derived from oblique imagery using machine learning.

The focus of this chapter was to assess the accuracy of a new end-to-end workflow for producing landcover data from oblique imagery. The workflow features two recently developed custom software tools: a trainable segmentation network and automated landcover classification algorithm, and a web-based georeferencing tool. Nineteen images from the Mountain Legacy Project (MLP) were classified and georeferenced using this workflow. These data were then combined using the mode value to produce a landcover map for a study area in Jasper NP. Mean georeferencing accuracy for a subset of seven images was 4.6 m, and the results were unaffected by distance from camera or angle of incidence.

Both the oblique and conventional landcover maps effectively depicted key landscape features in the study area such as the Athabasca River and other water bodies, the central herbaceous/shrub areas, and patches of broadleaf/mixedwood in the northern extent. However, the appearance of the two maps is noticeably different: the oblique landcover map is smoother in appearance whereas the orthogonal land cover map has the classic speckled appearance of a pixel-based classification. These differences are worth considering for future analyses that aim to quantify changes in chronological data derived from different sources

The oblique landcover map achieved 68% overall accuracy, outperforming the orthogonal landcover map's overall accuracy of 60%. Water, coniferous forest, and broadleaf/mixedwood classes scored well for the oblique landcover, but wetland and barren ground performed poorly due to class confusion with other landcover types. Comparing landcover class distribution between the two maps, barren ground and water shared similar proportions but there were large differences in the proportions of the other four landcover classes. This can be largely explained by the lower accuracy scores of the orthogonal map, specifically the considerable confusion between coniferous forest and broadleaf/mixedwood classes.

5.1.2 Chapter 3: A tale of two disturbances: Can mountain pine beetle restore landcover composition and pattern altered by fire suppression in Jasper National Park?

In Jasper NP, fire suppression over the previous century has altered the fire return interval, resulting in a significant increase of homogenous forest cover. These changes contributed to an unprecedented mountain pine beetle (MPB) outbreak over the previous decade (there is no prior occurrence of MPB in the park), which impacted up to 70% of lodgepole pine. In this chapter, I compared the composition and pattern of landcover following stand-replacing fires in 1889 and 1905, and the recent MPB outbreak, using the same data sources and workflow developed in the previous chapter.

The results indicate that the predominant landcover class in the 1915 landcover map was regenerating conifer (56.4 %), followed by mature conifer (20.1 %). Regenerating conifer also had the highest patch complexity and mean patch size. Prior to outbreak, contemporary landcover in the study area was dominated by the mature conifer class (82.7 %), which subsequently had the highest patch complexity and mean patch size. The overall accuracy of the contemporary landcover map was 79 %.

A change analysis revealed that 64.2 % of the historical landscape transitioned to mature conifer cover in 2020/22. Most new mature conifer cover originated from regenerating conifer and shrub. Following the outbreak, MPB-affected conifer comprised 28.9% of the study area. This change occurred at the expense of mature conifer, which decreased to 53.8 %. However, mature conifer still had the highest patch complexity and mean patch size.

This analysis suggests that the unprecedented MPB outbreak can restore heterogeneity to the landscape that was previously lost due to the effects of fire suppression. However, successional responses in MPB-impacted stands and the timing of future disturbance could lead to species shifts or transitions to non-forest. Coupled with climate change–driven conditions, these potential transitions could increase the departure of montane ecosystems from their historical range of variability (HRV), which may require shifts in how they are managed by Parks Canada.

5.1.3 Chapter 4: Beyond history: Anticipating the influence of a novel disturbance regime on future landcover change in Jasper National Park.

The altered fire regime and unprecedented MPB outbreak in Jasper NP represent a significant departure from ecological processes that have occurred historically. The emergence of novel disturbance raises concerns about the capacity of ecosystems to respond to these changes and the capability of management interventions to support resistance and resilience (Turner 2010; Turner & Seidl 2023). The purpose of this chapter was to undertake a scenario planning exercise to anticipate potential ecological changes that may occur in the park due to novel disturbance. We followed a qualitative approach outlined by Peterson et al. (2003) to develop four scenarios that describe plausible ecological outcomes in the montane ecoregion based on different combinations of analog and novel responses (to recent disturbances) and differing levels of management intervention.

Scenario 1 – Wishful thinking

Montane ecosystems successfully recover from recent disturbance events and the landscape returns to 20th century homogenous forest cover dominated by lodgepole pine. Management actions under this scenario would be limited and focus primarily on suppressing fires in high priority areas of the park.

Scenario 2 – Nostalgic naturalness

Intensive management actions are undertaken to return ecosystems to their HRV and increase heterogeneity across the landscape. Integral to this scenario is restoring the effects of the historical fire regime through the extensive use of commercial logging, mechanical thinning, brush clearing and prescribed fire. Once these actions are completed, managers can allow future wildfires to burn when conditions permit to maintain HRV in the montane ecoregion

Scenario 3 – Apocalypse novel

Ecological responses to the recent MPB outbreak and wildfires result in fundamental change in the dominant cover in the montane ecoregion, characterized by shifts to new species assemblages or alternative stable states (i.e., non-forest). Climate-driven novel disturbances occur more frequently and trigger further ecological transformations. The relative lack of management action under this scenario is a consequence of falling revenues in the park and public resistance to broad-scale ecological interventions.

Scenario 4 – Controlled chaos

As ecological novelty increases in the montane ecoregion, aggressive management intervention (e.g., >10,000 ha of tree-planting and prescribed fire) is used to maintain important ecosystem functions and services. Managers also undertake significant measures to protect resources from future novel disturbance by promoting non-forest species adjacent to critical infrastructure, dramatically increasing fire roads, fire breaks, and firefighting units and associated equipment, and introducing sophisticated surveillance and rapid response technologies.

5.2 Benefits and lessons learned

5.2.1 Systematic coverage of historical photographs.

The systematic approach used for capturing the historical survey images greatly benefited the research undertaken in this dissertation. By establishing camera stations on numerous peaks and promontories adjacent to Jasper NP's broad montane valleys, image coverage for areas of interest was excellent. In Chapter 2, 19 images covered 99.3% of the study area (the overlapping image coverage greatly improved mapping accuracy). In Chapter 3, 26 images covered 97.1% of the study area. Additionally, areas of the landscape that were obscured by landform features or forest in individual images were often filled in by images from adjacent camera stations or from stations across the valley. Hence, the systematic capture of the survey images is beneficial for producing historical and repeat landcover maps.

5.2.2 Ancillary landcover data from lower elevation photographs.

Another important benefit of the original survey design is the difference in elevation of the various camera stations. In Chapter 3, the average height of the 14 camera stations used in the analysis was 2,375 m, whereas the average height of the study area was 1,204 m. This difference in elevation, in combination with the distance from camera, can make it difficult to accurately classify similar landcover types in certain areas of the images. However, images from lower elevations, that are closer to areas of interest, can provide additional landcover information to correct classification errors. For example, images from several lower elevation stations (e.g., Power House Cliff at 1,160 m), that were excluded from the analysis because of their small area of study area coverage, still played an important role in differentiating between landcover classes (such as herbaceous and shrub or coniferous forest and broadleaf/mixedwood) visible on more distant, higher elevation images that were included in the analysis.

5.2.3 Accuracy of the georeferencing tool.

The georeferencing tool used in the end-to-end workflow produced results with significant positional accuracy for the work produced in this dissertation. In Chapter 2, the mean displacement for the subset of 7 images (i.e., one image per camera station) was 3.7m. This is especially impressive as the average distance from the camera to test points was 5,832.6 m. There is only one published study (see Bayr 2021) that has produced better accuracy results for georeferencing oblique photographs, where the average distance between camera and test points was 701 m (and the average height difference was 78 m).

There were two important factors for producing high georeferencing accuracy. First, the detail of the virtual photographs, which is dictated by the resolution of the digital surface models (DSM) from which they are constructed, greatly aids the identification of control points. DSMs include all features on the landscape (as opposed to digital terrain models, which only represent landform features), so the base of trees can be used as georeferencing points. DSMs are especially useful for georeferencing historical images because many of the trees visible in the virtual photographs derived from modern elevation data may also be present in the historical images, albeit shorter (hence using the base of the tree for control points). Second, the Image Analysis Toolkit enables smooth navigation for identifying features and the precise placement of control points.

5.2.4 Production of quantifiable landcover data.

The end-to-end workflow described in Chapter 2 enabled the production of georeferenced landcover maps derived from historical and repeated oblique photographs. The ability to work with these data in a GIS (or other spatial software applications) allows for conventional types of spatial analyses, such as producing area metrics for landcover classes, detailed change analyses, habitat modelling and patch metrics. For example, in Chapter 3, Fragstats 4.2 was used to produce patch metrics for the nine landcover classes in the historical and contemporary maps derived from oblique photographs (McGarigal et al. 2012). Thus, the new workflow constitutes an important advancement of methods for repeat photography analyses and builds on the work of previous MLP studies (see Fortin et al. 2019; Sanseverino et al. 2016; Rhemtulla et al. 2002).

5.2.5 Disturbance novelty is driving rapid change in Jasper NP.

The gradual then rapid landcover changes occurring in Jasper NP are the result of emergent disturbance novelty – a new term introduced by Turner and Seidl (2023) to describe disturbance events and regimes to which ecosystems are not adapted. Following a century of gradual change in land cover composition, the MPB outbreak from 1999 to 2018 and the Jasper Wildfire Complex (JWC) in 2024 have led to rapid unprecedented change. Given the influence of climate change and invasive species, there are valid concerns that ecosystem responses may fall outside their HRV (i.e., new species assemblages or shifts to non-forest species).

5.2.6 Scenarios can help navigate uncertainties.

There is considerable uncertainty about how the montane ecoregion will respond or recover from novel disturbance. Novel responses to novel disturbances could cause ecosystems to depart from their HRV, and transition to alternative stable states. Following the Canadian National Parks Act (2000), Jasper NP is mandated to maintain ecological integrity in the park. Profound ecological changes resulting from novel disturbances could challenge traditional management actions to restore historical processes and landcover configurations. Scenarios can help to navigate the uncertainty of ecological outcomes through continuous learning and modification. Monitoring systems, such as the ecological integrity monitoring program, are a key component of this approach. Managers evaluate monitoring data to track progress of performance measures. These data and other new information can then be used to adjust management actions or policies as necessary (Keenleyside et al. 2012).

5.3 Limitations

5.3.1 Classification issues related to grayscale images.

The classification performance of PyLC for historical grayscale images is below the standard of results produced for modern colour images (top performing models produced overall weighted F1 scores of 0.841 for grayscale images and 0.909 for color images). This may be due to several issues related to model training, including the limited spectral information of grayscale images and poor-quality images. The MLP is undertaking new work to address these issues. New deep learning models are being tested to improve classification performance for grayscale images. Additional tests involve improvements to image quality via software editing procedures (i.e., adjustments to the clarity, contrast and brightness of poor-quality images) and data augmentation. Finally, the original manual classifications used to train PyLC are being reassessed for accuracy and consistency and will be corrected as necessary to improve classification performance.

Another issue concerns the accuracy assessment of georeferenced classifications for historical images. This is because of limited orthogonal reference data (the historical survey images typically preceded aerial imagery by decades). Although assessment of the georeferenced classifications of the repeated images generally serves as a good proxy for the accuracy of historical images (as they have followed the same processing steps), care should be undertaken to ensure the quality of historical classifications match their modern counterparts when considering accuracy assessment.

Classification of landcover features further away from the camera can be challenging, especially for cover types that are difficult to distinguish between, such as mature conifer and broadleaf/mixedwood or shrub, wetland herbaceous cover. Generally, landcover in images used for Chapter 3 were not classified beyond a distance of 7 km from the camera. In instances where this distance was extended, adjacent images were used to confirm landcover furthest from the camera (see 5.2.1).

5.3.2 Manual corrections.

The landcover classifications produced by PyLC for grayscale historical and colour repeat images in Chapter 3 required time-consuming manual corrections (Appendix F). This process was undertaken using Affinity Photo Version 2.1, but numerous other photo editing software applications may be suitable depending on budget or user familiarity. Corrections involved developing a color palette that matched the landcover classes produced by PyLC, producing a grid overlay for tracking correction progress, and the use of editing tools to undertake the corrections. For Chapter 3, the time for correcting classifications ranged from 2-8 hours per image, depending on quality of the PyLC output (grayscale classifications usually required more corrections) or the complexity of the landcover captured in the image (e.g., the benchlands west of the town of Jasper required extensive corrections). An important benefit of producing the corrected classifications is these data can be added to the training dataset to improve PyLC accuracy and robustness.

5.3.3 New landcover categories need to be added manually.

PyLC was trained on two manually classified datasets that used 8- and 10- category classification schemes respectively, which were based on broad-level habitats found in the Canadian Rockies (also see Fisher et al. 2011; Jean et al. 2015). These categories excluded anthropogenic features and other broad landcover or disturbance categories found in Jasper NP. Without suitable training data, PyLC cannot classify new categories. Therefore, in Chapter 3, anthropogenic features were added to the barren ground category and new categories for herbaceous cover, regenerating conifer, and MPB-affected conifer required manual delineation. Again, these corrected classifications can be repurposed as training data to allow PyLC to classify these categories.

5.3.4 Poor classification performance for specific categories.

Certain landcover categories are difficult to classify (both manually and by PyLC). A good example are wetlands (in both grayscale and colour images), which are prone to changes in appearance through spring and summer months and are easily confused with other categories such as water, barren ground (owing to reflectiveness of standing water) or herbaceous cover. In Chapter 2, where no manual corrections were undertaken for the PyLC classifications, this category was frequently misclassified (Table 2.3). Previous studies acknowledge the challenge of correctly classifying certain landcover categories (e.g., Mahdavi et al. 2018).

5.3.5 Georeferencing historical images.

Accurate georeferencing requires identifying reciprocal control points in each quadrant of an image and virtual photograph. If there is an insufficient spread of control points, the transformation process produces warped images that don't align with the virtual photograph. Several historical and repeat images in Chapter 3 lacked discernible features for control points, especially for heavily forested lower quadrants. Workarounds are possible, such as trial and error control point placement using grid lines aligned to prominent features in other quadrants. However, this process is time-consuming and may require many iterations of control points to achieve satisfactory alignment. Future work to address this issue may involve the adaption of alignment applications that automate this process (see Okamoto et al. 2023).

5.3.6 Availability of high-resolution elevation data.

Another important requirement for producing accurate georeferencing results is the availability of high-resolution elevation datasets. Digital surface models (DSMs) are used for producing the virtual photographs, as they include above-ground features such as vegetation and human infrastructure. These features can serve as valuable control points, especially for the lower quadrants of images, when other topographic features aren't available. Digital terrain models (DTMs) are then required for georeferencing the classified data (the georeferenced data will be speckled or incomplete if surface models are used for this task). The availability and resolution of these data are important for producing accurate georeferencing results. The landcover maps produced for Chapters 2 and 3 used 2 m resolution elevation data (both DSMs and DTMs). The lower elevations of the study area comprised data from Natural Resources Canada. However, these data products are unavailable for higher elevations in the park. Further, the coverage for these data is limited outside of National

Parks in British Columbia and Alberta. Fortunately, both surface and terrain models can be derived from LiDAR data if available for areas of interest that are not covered by Natural Resources Canada. However, if neither dataset is available, georeferencing could be achieved using coarser elevation data but this would greatly impact georeferencing accuracy.

5.3.7 Compatibility with other data products.

An important consideration for producing landcover data from oblique images using the new workflow described in this dissertation is their incompatibility with other datasets produced from orthogonal imagery using conventional remote sensing interpretation techniques. Data derived from oblique and orthogonal imagery could potentially be used together to produce metrics for certain landcover categories in time sequences (with explicitly mentioned caveats explaining differences in data and interpretation methods) but spatial change analyses would be more challenging given the significant differences in appearance of these data products (i.e., Fig 2.4).

5.3.8 Software limitations.

Numerous software limitations have emerged from this research. As with many software applications developed for research purposes, the initial development focus was to support analysis objectives and produce usable results. Therefore, PyLC lacked a graphical user interface for the analysis undertaken in this dissertation and required implementing Python scripts to produce classifications. Training models on new data or introducing new models requires advanced programming knowledge.

Several models, each trained and optimized using different hyperparameters (i.e., variables that control the training process) are available for use in PyLC. Rose (2020) reported on the top-performing models for grayscale and colour images respectively but also acknowledged that model accuracy could be inconsistent and was highly dependant on the image input. Therefore, selecting the best model to produce an image classification is a time-consuming process and involves generating and visually assessing output from five models to select the best classification.

The georeferencing tool Is highly sensitive to incorrect data formatting. Elevation data need to be formatted to specific requirements, or the application will crash. This process can be especially challenging when combining different elevation datasets together, as required for the analysis undertaken in Chapters 2 and 3. Additionally, there are limits to the size of elevation datasets that can be used by the tool. For Chapter 3, this issue required splitting an elevation grid into smaller, overlapping sizes to georeference all images in the study area.

Finally, producing accurately georeferenced data is a time-consuming process due to limitations of the georeferencing tool. It requires a trial-and-error approach to fine tune the camera coordinates for producing a usable virtual photograph. A similar approach is also used for aligning the image – many iterations of control points may be required to produce a satisfactory alignment. Further, both tasks require the use of two separate software applications, requiring the user to toggle through windows and reopen datasets.

5.4. What might have been done differently.

5.4.1 Expanding the study area.

Initially, the study area for Chapter 3 included all three montane valleys in Jasper NP. However, after undertaking fieldwork to capture the required repeat photographs, correcting the historical and repeat classifications produced by PyLC required more time than anticipated. A decision was made to exclude one of the valleys from the analysis – the upper portion of the Athabasca River valley. However, after completing the analysis for Chapter 3 at the end of 2023, the following year the JWC would burn through most of this valley. Considering the uncertain recovery of the burned area, it would have been interesting to have made these data available for new change analyses.

5.4.2 Stakeholder engagement for scenario planning.

The scenario planning exercise described in Chapter 4 is focused on uncertainties related to potential outcomes of novel disturbance events. The intent of this exercise is to encourage managers, community members, and stakeholders to consider resilient conservation policies that anticipate the occurrence of these scenarios. A full scenario exercise would have involved a systematic consultation process. Given the shocking outcomes of the JWC in 2024 a study of this kind seemed beyond practical consideration. Engaging managers and scientists in scenario development would have improved the strength of the scenarios. Going forward, the scenarios I developed could be deployed in social processes to refine and test the scenarios in a real-world application

5.5 Recommendations

5.5.1 Further software development.

There is clear need to refine and package the tools tested and used in Chapters 2 and 3 into a standalone software application that integrates with a GIS. Fortunately, researchers associated with the MLP have recently released the Mountain Image Analysis Suite (MIAS; Wright et al. 2024), which immediately addresses several of the workflow limitations identified above in the Limitations section. MIAS is a freely available plugin developed for integration with QGIS – an open-source GIS application. MIAS provides a simple user interface for the workflow tools, allowing the user to input a grayscale or colour oblique image and produce a georeferenced classification following a series of easy-to-follow prompts and tasks. The user is also required to provide camera metadata (coordinates; azimuth and FOV angle) and elevation data. MIAS also incorporates many improvements to the georeferencing tool, such as enabling easy adjustments to refine the camera metadata, the selection of more than 4 alignment points (random sample consensus will remove outliers to select the four best points), and report on alignment point accuracy using RSME. Since the release of MIAS, a new tool has been added to the plugin that automatically combines overlapping georeferenced classifications using the process outlined in Chapters 2 and 3.

5.5.2 Improve PyLC performance.

In Chapter 2, the landcover map derived from oblique photographs classified by PyLC produced promising accuracy results (Table 2.3). Although individual images were all impacted by classification errors, the process of combining the georeferenced classifications improved accuracy (Fig. 2.6). In Chapter 3, both the historical and repeat classifications required extensive and time-consuming corrections (which did improve the accuracy on the contemporary landcover map). Despite the promise of PyLC, it is limited by its relatively small training dataset. Further, these data are derived from specific areas within the Canadian Rockies and are not wholly representative of the landscapes found throughout the range. To improve the accuracy and robustness, the original, manually produced training data can be reviewed and corrected. Additionally, new data from other regions in the Canadian Rockies and beyond may improve the robustness of PyLC. MLP researchers are currently undertaking this work with the intent of retraining PyLC to improve accuracy performance (see Mahindrakar et al. Submitted).

5.5.3 Continued repeat photography can benefit ecological monitoring.

Continued, periodic repeat photography of the 1915 survey photographs captured by M.P. Bridgland in Jasper NP has many advantages. To begin with, the original repeat photographs captured by Rhemtulla and Higgs between 1996-1998 are the earliest repeat photographs in the MLP collection. These "old" repeat photographs already serve as a valuable baseline for assessing recent changes in the park. The research undertaken in this dissertation involved a second repeat of camera stations overlooking the three major valleys in the park. Together, these three datasets are valuable for establishing baselines for monitoring purposes and tracking ecological responses to recent novel disturbance events in the park. For example, in Waterton Lakes National Park, following the 2017 Kenow Fire, repeat photography (of historical survey photographs) is being used as a tool for monitoring ecological recovery in the park. Given the considerable extent of the JWC, and the proximity to nearby camera stations, repeat photographs could serve as an effective ecological monitoring tool in Jasper NP. Additionally, the oblique angle of the photographs provides a unique perspective of the landscape and capture information that may not be available in orthogonal imagery (Fortin et al. 2019). This perspective (vs orthogonal imagery) may also be helpful for developing easy-to-follow resources for visitor interpretation regarding recent disturbance in the park.

5.5.4 Scenario planning can support management planning.

Novel disturbance regimes can compromise ecological integrity in National Parks by causing ecosystems to depart from their HRV. The scenario planning exercise undertaken in Chapter 4 evaluates the ecological consequences of emergent novel disturbance in Jasper NP. The intent of this exercise is to better prepare natural resource managers for potential ecological changes that threaten ecological integrity. Future planning efforts can anticipate these scenarios by developing new monitoring protocols for areas that are sensitive or exposed to novel disturbance. However, traditional management actions to restore historical processes and landcover configurations that are impacted by novel disturbance may be practically impossible at worst and ineffective at best. Therefore, scenario planning can compliment adaptive management strategies and enable actions that can mitigate undesired effects of novel disturbance, align management goals with emerging hybrid and novel ecosystems, and foster more resilient landscapes.

5.5.5 The continuing value of HRV.

The HRV concept was developed as a tool for understanding the causes and consequences of change in ecosystem conditions and processes over time (Morgan et al. 1994). It can describe variation in diverse characteristics (i.e., species densities, disturbance frequency or rates of change) and can be applied at multiple scales (Morgan et al. 1994). Natural resource managers have used HRV frameworks since the early 1960's to maintain biodiversity, restore degraded ecosystems, and serve as benchmarks for assessing anthropogenic change (Landres et al. 1999).

However, in an era of rapid directional change, where ecosystems are transitioning to irreversible alternative states (i.e., novel ecosystems), is the HRV concept still a relevant conservation tool? Higgs et al. (2014) suggest that the value of HRV to restoration ecology may diminish over time. Further, Duncan et al. (2010) argue that natural resource management requires anticipating the future rather than maintaining HRV. Nonetheless, HRV is still an important concept for establishing baselines for assessing ecological change and/or monitoring the emergence of novelty. HRV can also improve our understanding of the consequences of changes in ecosystem characteristics (Morgan et al. 1994).

The development of new techniques focused on increasing our knowledge of how historical ecosystems functioned and responded to change remains important for natural resource management. New research should also consider the value of the potential future range of variability (Turner & Seidl 2023). For example, simulation modeling can be used to determine the sensitivity of disturbance regimes to changing climate or identify ecological thresholds that could lead to regime shifts (Albrich et al. 2020; Turner et al. 2022). Further, with the availability of large, remotely sensed datasets, machine learning can play an important role in detecting change patterns or investigating underlying ecosystem mechanisms (Turner & Seidl 2023).

5.5.6 The challenge of maintaining ecological integrity.

Parks Canada's primary mandate in National Parks is to maintain ecological integrity (i.e., "a condition that is determined to be characteristic of its natural region"; Parks Canada 2024a). However, emergent novel disturbances may prompt ecological responses that result in fundamental ecological change (i.e., regime shifts; Turner & Seidl 2023). Efforts to restore the integrity of ecosystems undergoing rapid change may ultimately be misguided due to the irreversibility of ecosystem shifts (e.g., due to the influence of climate change on system hysteresis). This issue brings into question whether a) the ecological integrity concept is still an appropriate mandate for park managers, b) the concept can be redefined to remove explicit links to natural range of variation, or c) ecological integrity is tied to shifts in regional ecological change that allows for shifting local priorities? Wurtzebach and Schultz (2016) acknowledge that maintaining ecological integrity may be difficult in certain instances where ecological novelty (i.e., hybrid or novel ecosystems) is inevitable but also argue that the concept remains a useful framework for ensuring the scientific credibility and social relevance of targets for management and monitoring. Therefore, a redefined interpretation of ecological integrity could emphasize important ecosystem functions and services as well as the persistence of specific species (also see Lemieux et al. 2011).

References

Abatzoglou, J. T., & Williams, A. P. (2016). Impact of anthropogenic climate change on wildfire across western US forests. *Proceedings of the National Academy of Sciences*, *113*(42), 11770–11775. https://doi.org/10.1073/pnas.1607171113

Andersen, T., Carstensen, J., Hernández-García, E., & Duarte, C. M. (2009). Ecological thresholds and regime shifts: Approaches to identification. *Trends in Ecology & Evolution*, *24*(1), 49–57. https://doi.org/10.1016/j.tree.2008.07.014

Anderson-Teixeira, K. J., Miller, A. D., Mohan, J. E., Hudiburg, T. W., Duval, B. D., & DeLucia, E. H. (2013). Altered dynamics of forest recovery under a changing climate. *Global Change Biology*, *19*(7), 2001–2021. https://doi.org/10.1111/gcb.12194

Albrich, K., Rammer, W., & Seidl, R. (2020). Climate change causes critical transitions and irreversible alterations of mountain forests. *Global Change Biology*, *26*(7), 4013–4027. https://doi.org/10.1111/gcb.15118

Aplet, G. H & Cole, D.N. (2010). Planning in the context of uncertainty: flexibility for adapting to change. In: Cole, D.N., Yung, L., eds. Beyond naturalness: rethinking park and wilderness stewardship in an era of rapid change. Washington, DC: Island Press: 12-29.

Aplet, G. H., & McKinley, P. S. (2017). A portfolio approach to managing ecological risks of global change. *Ecosystem Health and Sustainability*, *3*(2), e01261. https://doi.org/10.1002/ehs2.1261

Astrup, R., Coates, K. D., & Hall, E. (2008). Recruitment limitation in forests: Lessons from an unprecedented mountain pine beetle epidemic. *Forest Ecology and Management*, *256*(10), 1743–1750. https://doi.org/10.1016/j.foreco.2008.07.025

Axelson, J. N., Alfaro, R. I., & Hawkes, B. C. (2009). Influence of fire and mountain pine beetle on the dynamics of lodgepole pine stands in British Columbia, Canada. *Forest Ecology and Management*, *257*(9), 1874–1882. https://doi.org/10.1016/j.foreco.2009.01.047

Axelson, J. N., Hawkes, B. C., vanAkker, L., & Alfaro, R. I. (2018). *Stand dynamics and the mountain pine beetle—30 years of forest change in Waterton Lakes National Park, Alberta, Canada*. 1159–1170. https://doi.org/10.1139/cjfr-2018-0161

Balch, J. K., Iglesias, V., Mahood, A. L., Cook, M. C., Amaral, C., DeCastro, A., Leyk, S., McIntosh, T. L., Nagy, R. C., St. Denis, L., Tuff, T., Verleye, E., Williams, A. P., & Kolden, C. A. (2024). The fastest-growing and most destructive fires in the US (2001 to 2020). *Science*, *386*(6720), 425–431. https://doi.org/10.1126/science.adk5737

Balch, J. K., & Williams, A. P. (2024). Extreme fire seasons are looming - Science can help us adapt. *Nature*, *634*(8036), 1041–1044. https://doi.org/10.1038/d41586-024-03433-y

Bassil, S., Froese, R. E., & Pinno, B. D. (2025). Little recovery of the residual stand after mountain pine beetle disturbance in old stands in the northern Rocky Mountains, Alberta, Canada. *Forest Ecology and Management*, *576*, 122408. https://doi.org/10.1016/j.foreco.2024.122408

Bayr, U. (2021) Quantifying historical landscape change with repeat photography: an accuracy assessment of geospatial data obtained through monoplotting. *International Journal of Geographical Information Science*, 35(10), 2026–2046. https://doi.org/10.1080/13658816.2021.1871910

Bayr, U., & Puschmann, O. (2019). Automatic detection of woody vegetation in repeat landscape photographs using a convolutional neural network. *Ecological Informatics*, *50*, 220–233. https://doi.org/10.1016/j.ecoinf.2019.01.012

Bedia, J., Herrera, S., Gutiérrez, J. M., Benali, A., Brands, S., Mota, B., & Moreno, J. M. (2015). Global patterns in the sensitivity of burned area to fire-weather: Implications for climate change. *Agricultural and Forest Meteorology*, *214–215*, 369–379. https://doi.org/10.1016/j.agrformet.2015.09.002

Beisner, B. E., Haydon, D. T., & Cuddington, K. (2003). Alternative stable states in ecology. *Frontiers in Ecology and the Environment*, *1*(7), 376–382. https://doi.org/10.1890/1540-9295(2003)001[0376:ASSIE]2.0.CO;2

Bentz, B. J., Duncan, J. P., & Powell, J. A. (2016). Elevational shifts in thermal suitability for mountain pine beetle population growth in a changing climate. *Forestry: An International Journal of Forest Research*, *89*(3), 271–283. https://doi.org/10.1093/forestry/cpv054

Beschta, R. L., & Ripple, W. J. (2007). Wolves, elk, and aspen in the winter range of Jasper National Park, Canada. *Canadian Journal of Forest Research*, *37*(10), 1873–1885. https://doi.org/10.1139/X07-017

Bozzini, C., Conedera, M., & Krebs, P. (2012). A New Monoplotting Tool to Extract Georeferenced Vector Data and Orthorectified Raster Data from Oblique Non-Metric Photographs. *International Journal of Heritage in the Digital Era*, *1*(3), 499–518. https://doi.org/10.1260/2047-4970.1.3.499

Bradley, M., & Neufeld, L. (2012). Climate and management interact to explain the decline of woodland caribou (Rangifer tarandus caribou) in Jasper National Park. *Rangifer*, 183–191. https://doi.org/10.7557/2.32.2.2268

Brown J.K. (1975). Fire cycles and community dynamics in lodgepole pine forests. In: D.M.Baumgartner . editor. Symposium Proceedings: Management of lodgepole pine ecosystems. Washington State University Cooperative Extension Service. Pullman, Washington, USA 430–456. https://doi.org/10.5555/19760634016

Buch, J., Williams, A. P., Juang, C. S., Hansen, W. D., & Gentine, P. (2023). SMLFire1.0: A stochastic machine learning (SML) model for wildfire activity in the western United States. *Geoscientific Model Development*, *16*(12), 3407–3433. https://doi.org/10.5194/gmd-16-3407-2023

Buma, B. (2015). Disturbance interactions: Characterization, prediction, and the potential for cascading effects. *Ecosphere*, *6*(4), art70. https://doi.org/10.1890/ES15-00058.1

Buma, B., Brown, C. D., Donato, D. C., Fontaine, J. B., & Johnstone, J. F. (2013). The Impacts of Changing Disturbance Regimes on Serotinous Plant Populations and Communities. *BioScience*, *63*(11), 866–876. https://doi.org/10.1525/bio.2013.63.11.5

Buma, B., & Wessman, C. A. (2012). Differential species responses to compounded perturbations and implications for landscape heterogeneity and resilience. *Forest Ecology and Management*, *266*, 25–33. https://doi.org/10.1016/j.foreco.2011.10.040

Burke, J., Bohlmann, J., & L. Carroll, A. (2017). Consequences of distributional asymmetry in a warming environment: Invasion of novel forests by the mountain pine beetle. *Ecosphere*, *8*, e01778. https://doi.org/10.1002/ecs2.1778

Busby, S. U., Moffett, K. B., & Holz, A. (2020). High-severity and short-interval wildfires limit forest recovery in the Central Cascade Range. *Ecosphere*, *11*(9), e03247. https://doi.org/10.1002/ecs2.3247

Byers, A.C. (2007) An assessment of contemporary glacier fluctuations in Nepal's Khumbu Himal using repeat photography. *Himalayan Journal of Sciences*, 4(6), 21–26. https://doi.org/10.3126/hjs.v4i6.979

Capon, S. J., Lynch, A. J. J., Bond, N., Chessman, B. C., Davis, J., Davidson, N., Finlayson, M., Gell, P. A., Hohnberg, D., Humphrey, C., Kingsford, R. T., Nielsen, D., Thomson, J. R., Ward, K., & Nally, R. M. (2015). Regime shifts, thresholds and multiple stable states in freshwater ecosystems; a critical appraisal of the evidence. *Science of The Total Environment*, *534*, 122–130. https://doi.org/10.1016/j.scitotenv.2015.02.045

Canham C.D. (1989). Different responses to gaps among shade-tolerant tree species. Ecology 70(3):548–550. https://doi.org/10.2307/1940200

Carpenter, S. R., Bennett, E. M., & Peterson, G. D. (2006). Scenarios for Ecosystem Services: An Overview. *Ecology and Society*, *11*(1). https://www.jstor.org/stable/26267787

Carroll, A. L., Régnière, J., Logan, J. A., Taylor, S. W., Bentz, B. J., & Powell, J. A. (2006). *Impacts of Climate Change on Range Expansion by the Mountain Pine Beetle* (Vols. 2006–14). http://cfs.nrcan.gc.ca/publications?id=26601

Carroll, A. L., Taylor, S. W., Régnière, J., & Safranyik, L. (2004). *Effects of climate change on range expansion by the mountain pine beetle in British Columbia* (Vol. 399). http://cfs.nrcan.gc.ca/publications?id=25051

Chapin, S. F., Zavaleta, E. S., Welling, L. A., Deprey, P., & Yung, L. (2010). Planning in the context of uncertainty: flexibility for adapting to change. In: Cole, D.N., Yung, L., eds. Beyond naturalness: rethinking park and wilderness stewardship in an era of rapid change. Washington, DC: Island Press: 216-233.

Chavardès, R. D., & Daniels, L. D. (2016). Altered mixed-severity fire regime has homogenised montane forests of Jasper National Park. *International Journal of Wildland Fire*, *25*(4), 433. https://doi.org/10.1071/WF15048

Chen, L.-C., Papandreou, G., Kokkinos, I., Murphy, K. & Yuille, A.L. (2017) DeepLab: semantic image segmentation with deep convolutional nets, Atrous convolution, and fully connected CRFs. *IEEE Transactions on Pattern Analysis and Machine Intelligence*, 40(4), 834–848.

Christianson, A. (2014). Social science research on Indigenous wildfire management in the 21st century and future research needs. *International Journal of Wildland Fire*, *24*(2), 190–200. https://doi.org/10.1071/WF13048

Clark, P.E. & Hardegree, S.P. (2005) Quantifying vegetation change by point sampling landscape photography time series. *Rangeland Ecology & Management*, 58(6), 588–597. https://doi.org/10.2111/04-111R2.1

Cobb, R. C., & Metz, M. R. (2017). Tree Diseases as a Cause and Consequence of Interacting Forest Disturbances. *Forests*, *8*(5), Article 5. https://doi.org/10.3390/f8050147

Collins, L., Bradstock, R. A., Clarke, H., Clarke, M. F., Nolan, R. H., & Penman, T. D. (2021). The 2019/2020 mega-fires exposed Australian ecosystems to an unprecedented extent of high-severity fire. *Environmental Research Letters*, *16*(4), 044029. https://doi.org/10.1088/1748-9326/abeb9e

Cooke, B. J., & Carroll, A. L. (2017). Predicting the risk of mountain pine beetle spread to eastern pine forests: Considering uncertainty in uncertain times. *Forest Ecology and Management*, *396*, 11–25. https://doi.org/10.1016/j.foreco.2017.04.008

Coop, J. D., Parks, S. A., Stevens-Rumann, C. S., Crausbay, S. D., Higuera, P. E., Hurteau, M. D., Tepley, A., Whitman, E., Assal, T., Collins, B. M., Davis, K. T., Dobrowski, S., Falk, D. A., Fornwalt, P. J., Fulé, P. Z., Harvey, B. J., Kane, V. R., Littlefield, C. E., Margolis, E. Q., … Rodman, K. C. (2020). Wildfire-Driven Forest Conversion in Western North American Landscapes. *BioScience*, *70*(8), 659–673. https://doi.org/10.1093/biosci/biaa061

Coop, J. D., Parks, S. A., Stevens-Rumann, C. S., Ritter, S. M., & Hoffman, C. M. (2022). Extreme fire spread events and area burned under recent and future climate in the western USA. Global Ecology and Biogeography. 31: 1949-1959. https://doi.org/10.1111/geb.13496

Cooper, G. S., Willcock, S., & Dearing, J. A. (2020). Regime shifts occur disproportionately faster in larger ecosystems. Nature Communications, 11(1), 1175. https://doi.org/10.1038/s41467-020-15029-x

Craig, R. K. (2010). “Stationarity is Dead” - Long Live Transformation: Five Principles for Climate Change Adaptation Law (SSRN Scholarly Paper 1357766). Social Science Research Network. https://papers.ssrn.com/abstract=1357766

Cudmore, T. J., Björklund, N., Carroll, A. L., & Staffan Lindgren, B. (2012). Climate change and range expansion of an aggressive bark beetle: Evidence of higher beetle reproduction in naïve host tree populations. Journal of Applied Ecology, 1036–1043. https://doi.org/10.1111/j.1365-2664.2010.01848.x@10.1111/(ISSN)1365-2664.CLIMATE_JPE

Cullingham, C. I., Cooke, J. E. K., Dang, S., Davis, C. S., Cooke, B. J., & Coltman, D. W. (2011). Mountain pine beetle host-range expansion threatens the boreal forest. Molecular Ecology, 20(10), 2157–2171. https://doi.org/10.1111/j.1365-294X.2011.05086.x

Dale, V. H., Joyce, L. A., McNulty, S., & Neilson, R. P. (2000). The interplay between climate change, forests, and disturbances. Science of The Total Environment, 262(3), 201–204. https://doi.org/10.1016/S0048-9697(00)00522-2

Dalman, D. 2004. Mountain pine beetle management in Canada's mountain national parks. In Mountain Pine Beetle Symposium: Challenges and Solutions, 30–31 October 2003, Kelowna, British Columbia. Edited by T.L. Shore, J.E. Brooks, and J.E. Stone. Can. For. Serv. Inf. Rep. BC-X-399. Available from http://www.for.gov.bc.ca/hfd/library/mpb/bib93473.pdf. pp. 87–93.

de la Giroday, H.-M. C., Carroll, A. L., & Aukema, B. H. (2012). Breach of the northern Rocky Mountain geoclimatic barrier: Initiation of range expansion by the mountain pine beetle. Journal of Biogeography, 39(6), 1112–1123. https://doi.org/10.1111/j.1365-2699.2011.02673.x

Dickson-Hoyle, S., Ignace, R. E., Ignace, M. B., Hagerman, S. M., Daniels, L. D., & Copes-Gerbitz, K. (2022). Walking on two legs: A pathway of Indigenous restoration and reconciliation in fire-adapted landscapes. Restoration Ecology, 30(4), e13566. https://doi.org/10.1111/rec.13566

Dordel, J., Feller, M. C., & Simard, S. W. (2008). Effects of mountain pine beetle (Dendroctonus ponderosae Hopkins) infestations on forest stand structure in the southern Canadian Rocky Mountains. *Forest Ecology and Management*, *255*(10), 3563–3570. https://doi.org/10.1016/j.foreco.2008.02.018

Dudney, J., D'Antonio, C., Hobbs, R. J., Shackelford, N., Standish, R. J., & Suding, K. N. (2024). Capacity for change: Three core attributes of adaptive capacity that bolster restoration efficacy. *Restoration Ecology*, *32*(8), e13647. https://doi.org/10.1111/rec.13647

Duncan, S. L., McComb, B. C., & Johnson, K. N. (2010). Integrating Ecological and Social Ranges of Variability in Conservation of Biodiversity: Past, Present, and Future. *Ecology and Society*, *15*(1). https://www.jstor.org/stable/26268091

ESRI. (2021) *ArcGIS pro: release 2.8.3.* Redlands, CA: Environmental Systems Research Institute.

ESRI (2022) ArcGIS pro: release 3.0.2. Environmental Systems Research Institute, Redlands, CA

Evers, C. R., Wardropper, C. B., Branoff, B., Granek, E. F., Hirsch, S. L., Link, T. E., Olivero-Lora, S., & Wilson, C. (2018). The ecosystem services and biodiversity of novel ecosystems: A literature review. *Global Ecology and Conservation*, *13*, e00362. https://doi.org/10.1016/j.gecco.2017.e00362

Fettig, C. J., Reid, M. L., Bentz, B. J., Sevanto, S., Spittlehouse, D. L., & Wang, T. (2013). Changing Climates, Changing Forests: A Western North American Perspective. *Journal of Forestry*, *111*(3), 214–228. https://doi.org/10.5849/jof.12-085

Fisher, J. T., Wheatley, M., & Gould, J. (2011). Rocky Mountian Biodiversity: Ecological communities and rare elusive species in heterogeneous landscapes (Interim Report), Willmore Biodiversity

Research Project. *Government of Alberta, Alberta Innovates-Technology Futures, Alberta Biodiveristy Monitoring Institute, Alberta*.

Folke, C., Carpenter, S., Walker, B., Scheffer, M., Elmqvist, T., Gunderson, L., & Holling, C. S. (2004). Regime Shifts, Resilience, and Biodiversity in Ecosystem Management. *Annual Review of Ecology, Evolution, and Systematics*, *35*(1), 557–581. https://doi.org/10.1146/annurev.ecolsys.35.021103.105711

Fortin, J. A., Fisher, J. T., Rhemtulla, J. M., & Higgs, E. S. (2019). Estimates of landscape composition from terrestrial oblique photographs suggest homogenization of Rocky Mountain landscapes over the last century. *Remote Sensing in Ecology and Conservation*, *5*(3), 224–236. https://doi.org/10.1002/rse2.100

García-Llamas, P., Suárez-Seoane, S., Taboada, A., Fernández-Manso, A., Quintano, C., Fernández-García, V., Fernández-Guisuraga, J. M., Marcos, E., & Calvo, L. (2019). Environmental drivers of fire severity in extreme fire events that affect Mediterranean pine forest ecosystems. Forest Ecology and Management, 433, 24–32. https://doi.org/10.1016/j.foreco.2018.10.051

Gat, C., Albu, A. B., German, D., & Higgs, E. (2011). A Comparative Evaluation of Feature Detectors on Historic Repeat Photography. In G. Bebis, R. Boyle, B. Parvin, D. Koracin, S. Wang, K. Kyungnam, B. Benes, K. Moreland, C. Borst, S. DiVerdi, C. Yi-Jen, & J. Ming (Eds.), *Advances in Visual Computing* (Vol. 6939, pp. 701–714). Springer Berlin Heidelberg. https://doi.org/10.1007/978-3-642-24031-7_70

Government of Alberta. (2024) Traffic volumes at points on the highway. Retrieved February 15, 2025, from https://open.alberta.ca/opendata/traffic-volumes-at-points-on-the-highway/resource/e20b7c7b-5580-4dc4-9339-c7f31c8d1716

Government of British Columbia (2022) Vegetation Resources Inventory. https://www2.gov.bc.ca/gov/content/industry/forestry/managing-our-forest-resources/forest-inventory/data-management-and-access

Green, K. (2000) Selecting and interpreting high-resolution images. *Journal of Forestry*, 98(6), 37–40. https://doi.org/10.1093/jof/98.6.37

Groffman, P. M., Baron, J. S., Blett, T., Gold, A. J., Goodman, I., Gunderson, L. H., Levinson, B. M., Palmer, M. A., Paerl, H. W., Peterson, G. D., Poff, N. L., Rejeski, D. W., Reynolds, J. F., Turner, M. G., Weathers, K. C., & Wiens, J. (2006). Ecological Thresholds: The Key to Successful Environmental Management or an Important Concept with No Practical Application? *Ecosystems*, *9*(1), 1–13. https://doi.org/10.1007/s10021-003-0142-z

Gruell, G.E. (1983) *Fire and vegetative trends in the northern Rockies: interpretations from 1871–1982 photographs.* General technical report INT-158. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station.

Gruell, G.E., Schmidt, W.C., Arno, S.F. & Reich, W.J. (1982) *Seventy years of vegetative change in a managed ponderosa pine Forest in Western Montana—implications for resource management*. Ogden, Utah: U.S. Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station.

Hagmann, R. K., Hessburg, P. F., Prichard, S. J., Povak, N. A., Brown, P. M., Fulé, P. Z., Keane, R. E., Knapp, E. E., Lydersen, J. M., Metlen, K. L., Reilly, M. J., Sánchez Meador, A. J., Stephens, S. L., Stevens, J. T., Taylor, A. H., Yocom, L. L., Battaglia, M. A., Churchill, D. J., Daniels, L. D., … Waltz, A. E. M. (2021). Evidence for widespread changes in the structure, composition, and fire regimes of western North American forests. *Ecological Applications*, *31*(8), e02431. https://doi.org/10.1002/eap.2431

Hallett, L. M., Standish, R. J., Hulvey, K. B., Gardener, M. R., Suding, K. N., Starzomski, B. M., Murphy, S. D., & Harris, J. A. (2013). Towards a Conceptual Framework for Novel Ecosystems. In Novel Ecosystems (pp. 16–28). John Wiley & Sons, Ltd. https://doi.org/10.1002/9781118354186.ch3

Harris, J. A., Hobbs, R. J., Higgs, E., & Aronson, J. (2006). Ecological Restoration and Global Climate Change. *Restoration Ecology*, *14*(2), 170–176. https://doi.org/10.1111/j.1526-100X.2006.00136.x

Harris, J. A., Murphy, S. D., Nelson, C. R., Perring, M. P., & Tognetti, P. M. (2013). Characterizing Novel Ecosystems: Challenges for Measurement. In *Novel Ecosystems* (pp. 192–204). John Wiley & Sons, Ltd. https://doi.org/10.1002/9781118354186.ch24

Harrison, G. W. (1979). Stability under Environmental Stress: Resistance, Resilience, Persistence, and Variability. *The American Naturalist*, *113*(5), 659–669. https://doi.org/10.1086/283424

Harvey, B. J., Donato, D. C., & Turner, M. G. (2016). High and dry: Post-fire tree seedling establishment in subalpine forests decreases with post-fire drought and large stand-replacing burn patches. Global Ecology and Biogeography, 25(6), 655–669. https://doi.org/10.1111/geb.12443

Hastings, J. R., & Turner, R. M. (1965). The changing mile. An ecological study of vegetation change with time in the lower mile of an arid and semiarid region. *The Changing Mile. An Ecological Study of Vegetation Change with Time in the Lower Mile of an Arid and Semiarid Region*. https://www.cabdirect.org/cabdirect/abstract/19670700517

Hayward, G.D., Veblen, T.T., Suring, L.H. & Davis, B. (2012) Challenges in the application of historical range of variation to conservation and land management. In: Wiens, J.A., Hayward, G.D., Safford, H.D. & Giffen, C.M. (Eds.) *Historical environmental variation in conservation and natural resource management.* Hoboken, NJ: Wiley-Blackwell, pp. 32–45.

He, T., Lamont, B. B., & Pausas, J. G. (2019). Fire as a key driver of Earth's biodiversity. Biological Reviews, 94(6), 1983–2010. https://doi.org/10.1111/brv.12544

Heger, T., Bernard-Verdier, M., Gessler, A., Greenwood, A. D., Grossart, H.-P., Hilker, M., Keinath, S., Kowarik, I., Kueffer, C., Marquard, E., Müller, J., Niemeier, S., Onandia, G., Petermann, J. S., Rillig, M. C., Rödel, M.-O., Saul, W.-C., Schittko, C., Tockner, K., … Jeschke, J. M. (2019). Towards an Integrative, Eco-Evolutionary Understanding of Ecological Novelty: Studying and Communicating Interlinked Effects of Global Change. *BioScience*. https://doi.org/10.1093/biosci/biz095

Hewitt, J., Thrush, S., Lohrer, A., & Townsend, M. (2010). A latent threat to biodiversity: Consequences of small-scale heterogeneity loss. Biodiversity and Conservation, 19(5), 1315–1323. https://doi.org/10.1007/s10531-009-9763-7

Higgs, E. (2003) *Nature by design: people, natural process, and ecological restoration.* Cambridge, MA: MIT Press.

Higgs, E., Falk, D. A., Guerrini, A., Hall, M., Harris, J., Hobbs, R. J., Jackson, S. T., Rhemtulla, J. M., & Throop, W. (2014). The changing role of history in restoration ecology. Frontiers in Ecology and the Environment, 12(9), 499–506. https://doi.org/10.1890/110267

Higgs, E. S., Sanseverino, M. E., Whitney, M. J., & Fortin, J. (2020). *Advances in Visual Applications: Visualizing & Quantifying Landscape Change in SW Alberta Using Mountain Legacy Project Photography (Unpublished report).* University of Victoria.

Hobbs, R. J., Higgs, E. S., & Hall, C. M. (2013). Defining Novel Ecosystems. In *Novel Ecosystems* (pp. 58–60). John Wiley & Sons, Ltd. https://doi.org/10.1002/9781118354186.ch6

Hobbs, R. J., Higgs, E., Hall, C. M., Bridgewater, P., Chapin, F. S., Ellis, E. C., Ewel, J. J., Hallett, L. M., Harris, J., Hulvey, K. B., Jackson, S. T., Kennedy, P. L., Kueffer, C., Lach, L., Lantz, T. C., Lugo, A. E., Mascaro, J., Murphy, S. D., Nelson, C. R., … Yung, L. (2014). Managing the whole landscape: Historical, hybrid, and novel ecosystems. Frontiers in Ecology and the Environment, 12(10), 557–564. https://doi.org/10.1890/130300

Hobbs, R. J., Higgs, E., & Harris, J. A. (2009). Novel ecosystems: Implications for conservation and restoration. Trends in Ecology & Evolution, 24(11), 599–605. https://doi.org/10.1016/j.tree.2009.05.012

Hoecker, T. J., & Turner, M. G. (2022). A short-interval reburn catalyzes departures from historical structure and composition in a mesic mixed-conifer forest. Forest Ecology and Management, 504, 119814. https://doi.org/10.1016/j.foreco.2021.119814

Hoffman, M.T. & Todd, S. (2010) Using fixed point photography, field surveys, and GIS to monitor environmental change: an example from Riemvasmaak, South Africa. In: Webb, R.H., Boyer, D.E. & Turner, R.M. (Eds.) *Repeat photography: methods and applications in the natural sciences.* Washington, DC: Island Press, pp. 46–56.

Hoffman, K. M., Christianson, A. C., Dickson-Hoyle, S., Copes-Gerbitz, K., Nikolakis, W., Diabo, D. A., McLeod, R., Michell, H. J., Mamun, A. A., Zahara, A., Mauro, N., Gilchrist, J., Ross, R. M., & Daniels, L. D. (2022). The right to burn: Barriers and opportunities for Indigenous-led fire stewardship in Canada. *FACETS*, *7*, 464–481. https://doi.org/10.1139/facets-2021-0062

Holland, W. D., & Coen, G. M. (1983). *Ecological (biophysical) land classification of Banff and Jasper National Parks . Volume I: Summary*. http://cfs.nrcan.gc.ca/publications?id=27562

Holling, C. S. (1973). Resilience and Stability of Ecological Systems. *Annual Review of Ecology and Systematics*, *4*(1), 1–23. https://doi.org/10.1146/annurev.es.04.110173.000245

Hong, S., Noh, H. & Han, B. (2015) Decoupled deep neural network for semi-supervised semantic segmentation (arXiv:1506.04924). *arXiv*. https://doi.org/10.48550/arXiv.1506.04924

Humphries, H. C., & Bourgeron, P. S. (2001). Methods for Determining Historical Range of Variability. In M. E. Jensen & P. S. Bourgeron (Eds.), *A Guidebook for Integrated Ecological Assessments* (pp. 273–291). Springer. https://doi.org/10.1007/978-1-4419-8620-7_20

Hung, W.-C., Tsai, Y.-H., Liou, Y.-T., Lin, Y.-Y. & Yang, M.-H. (2018) Adversarial learning for semi-supervised semantic segmentation (arXiv:1802.07934). *arXiv*. https://doi.org/10.48550/arXiv.1802.07934

Jasper National Park (2024). Jasper Wildfire Complex perimeter [Unpublished shapefile dataset]. Parks Canada

Jean, F., Albu, A. B., Capson, D., Higgs, E., Fisher, J. T., & Starzomski, B. M. (2015). The Mountain Habitats Segmentation and Change Detection Dataset. *2015 IEEE Winter Conference on Applications of Computer Vision*, 603–609. https://doi.org/10.1109/WACV.2015.86

Johnstone, J. F., Allen, C. D., Franklin, J. F., Frelich, L. E., Harvey, B. J., Higuera, P. E., Mack, M. C., Meentemeyer, R. K., Metz, M. R., Perry, G. L., Schoennagel, T., & Turner, M. G. (2016). Changing disturbance regimes, ecological memory, and forest resilience. Frontiers in Ecology and the Environment, 14(7), 369–378. https://doi.org/10.1002/fee.1311

Jolly, W. M., Cochrane, M. A., Freeborn, P. H., Holden, Z. A., Brown, T. J., Williamson, G. J., & Bowman, D. M. J. S. (2015). Climate-induced variations in global wildfire danger from 1979 to 2013. Nature Communications, 6(1), 7537–7537. https://doi.org/10.1038/ncomms8537
Jones, M. W., Abatzoglou, J. T., Veraverbeke, S., Andela, N., Lasslop, G., Forkel, M., Smith, A. J. P., Burton, C., Betts, R. A., van der Werf, G. R., Sitch, S., Canadell, J. G., Santín, C., Kolden, C., Doerr, S. H., & Le Quéré, C. (2022). Global and Regional Trends and Drivers of Fire Under Climate Change. *Reviews of Geophysics*, *60*(3), e2020RG000726. https://doi.org/10.1029/2020RG000726

Jones, M. W., Abatzoglou, J. T., Veraverbeke, S., Andela, N., Lasslop, G., Forkel, M., Smith, A. J. P., Burton, C., Betts, R. A., van der Werf, G. R., Sitch, S., Canadell, J. G., Santín, C., Kolden, C., Doerr, S. H., & Le Quéré, C. (2022). Global and Regional Trends and Drivers of Fire Under Climate Change. *Reviews of Geophysics*, *60*(3), e2020RG000726. https://doi.org/10.1029/2020RG000726

Joseph, M. B., Rossi, M. W., Mietkiewicz, N. P., Mahood, A. L., Cattau, M. E., Denis, L. A. S., Nagy, R. C., Iglesias, V., Abatzoglou, J. T., & Balch, J. K. (2019). Spatiotemporal prediction of wildfire size extremes with Bayesian finite sample maxima. *Ecological Applications*, *29*(6), e01898. https://doi.org/10.1002/eap.1898

Keane, R. (2013). Disturbance regimes and the historical range of variation in terrestrial ecosystems [Chapter 389]. *In: Levin, S. A., Ed. Encyclopedia of Biodiversity, Volume 2. Waltham, MA: Academic Press. p. 568-581.*, 568–581. https://doi.org/10.1016/B978-0-12-384719-5.00389-0

Keane, R. E., Hessburg, P. F., Landres, P. B., & Swanson, F. J. (2009). The use of historical range and variability (HRV) in landscape management. *Forest Ecology and Management*, *258*(7), 1025–1037. https://doi.org/10.1016/j.foreco.2009.05.035

Keane, R. E., Ryan, K. C., Veblen, T. T., Allen, C. D., Logan, J., & Hawkes, B. (2002). Cascading effects of fire exclusion in the Rocky Mountain ecosystems: A literature review. *General Technical Report. RMRS-GTR-91. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 24 p.*, *091*. https://doi.org/10.2737/RMRS-GTR-91

Keenleyside, K., Dudley, N., Cairns, S., Hall, C., & Stolton, S. (2012). *Ecological restoration for protected areas: Principles, guidelines and best practices*. IUCN. https://www.iucn.org/content/ecological-restoration-protected-areas-principles-guidelines-and-best-practices

Kerr, M. R., Ordonez, A., Riede, F., Atkinson, J., Pearce, E. A., Sykut, M., Trepel, J., & Svenning, J.-C. (2025). Widespread ecological novelty across the terrestrial biosphere. Nature Ecology & Evolution. https://doi.org/10.1038/s41559-025-02662-2

Krawchuk, M. A., Moritz, M. A., Parisien, M.-A., Dorn, J. V., & Hayhoe, K. (2009). Global Pyrogeography: The Current and Future Distribution of Wildfire. *PLoS ONE*, *4*(4), e5102. https://doi.org/10.1371/journal.pone.0005102

Kull, C.A. (2005) Historical landscape repeat photography as a tool for land use change research. *Norsk Geografisk Tidsskrift—Norwegian Journal of Geography*, 59(4), 253–268. Available from: https://doi.org/10.1080/00291950500375443

Kurz, W. A., Dymond, C. C., Stinson, G., Rampley, G. J., Neilson, E. T., Carroll, A. L., Ebata, T., & Safranyik, L. (2008). Mountain pine beetle and forest carbon feedback to climate change. Nature, 452(7190), 987–990. https://doi.org/10.1038/nature06777

Landres, P. B., Morgan, P., & Swanson, F. J. (1999). Overview of the Use of Natural Variability Concepts in Managing Ecological Systems. *Ecological Applications*, *9*(4), 1179–1188. https://doi.org/10.1890/1051-0761(1999)009[1179:OOTUON]2.0.CO;2

Lemieux, C. J., Beechey, T. J., & Gray, P. A. (2011). Prospects for Canada's protected areas in an era of rapid climate change. Land Use Policy, 28(4), 928–941. https://doi.org/10.1016/j.landusepol.2011.03.008

Lewis H.T., Ferguson T.A. (1988). Yards, corridors, and mosaics: how to burn a boreal forest. Hum Ecol 16(1):57–77. https://link.springer.com/article/10.1007/BF01262026

Lewontin, R.C. (1969). The meaning of stability. Brookhaven Symp Biol 22: 13–23.

Li, C., Flannigan, M. D., & Corns, I. G. (2000). Influence of potential climate change on forest landscape dynamics of west-central Alberta. Canadian Journal of Forest Research, 30(12), 1905–1912. https://doi.org/10.1139/x00-118

Lieffers, V. J., Benedik, J., Stadt, K., & Macdonald, S. E. (2023). Poor regeneration of pine after mountain pine beetle attack in colder boreal regions of Canada. Canadian Journal of Forest Research. https://doi.org/10.1139/cjfr-2023-0113

Lindenmayer, D. B., Hobbs, R. J., Likens, G. E., Krebs, C. J., & Banks, S. C. (2011). Newly discovered landscape traps produce regime shifts in wet forests. Proceedings of the National Academy of Sciences, 108(38), 15887–15891. https://doi.org/10.1073/pnas.1110245108

Luckman, B. H. (1998). Landscape and Climate Change in the Central Canadian Rockies During the 20th Century. *Canadian Geographies / Géographies Canadiennes*, *42*(4), 319–336. https://doi.org/10.1111/j.1541-0064.1998.tb01349.x

MacLaren, P.I.S., Higgs, E. & Zezulka-Mailloux, G. (2005) *Mapper of mountains: M.P. Bridgland in the Canadian Rockies, 1902–1930*, 1st edition. Edmonton, AB: The University of Alberta Press.

MacLaren, I. S., Payne, M., Murphy, P. J., Reichwein, P., McDermott, L., Taylor, C. J., Zezulka-Mailloux, G., Robinson, Z., & Higgs, E. (2012). *Culturing Wilderness in Jasper National Park: Studies in Two Centuries of Human History in the Upper Athabasca River Watershed*. University of Alberta Press. https://doi.org/10.1515/9780888645708

MacLaren, I. S., Payne, M., Murphy, P. J., Reichwein, P., McDermott, L., Taylor, C. J., Zezulka-Mailloux, G., Robinson, Z., Higgs, E., & Chrétien, T. R. H. J. (2007). "Following the Base of the Foothills" | Murphy. In *Culturing Wilderness in Jasper National Park*. The University of Alberta Press. MacLaren, P. I. S., Higgs, E., & Zezulka-Mailloux, G. (2005). *Mapper of Mountains: M.P. Bridgland in the Canadian Rockies, 1902-1930* (First edition edition). The University of Alberta Press.

Mahdavi, S., Salehi, B., Granger, J., Amani, M., Brisco, B. & Huang, W. (2018) Remote sensing for wetland classification: a comprehensive review. *GIScience & Remote Sensing*, 55(5), 623–658. https://doi.org/10.1080/15481603.2017.1419602

Masubelele, M.L., Hoffman, M.T. & Bond, W.J. (2015) A repeat photograph analysis of long-term vegetation change in semi-arid South Africa in response to land use and climate. *Journal of Vegetation Science*, 26(5), 1013–1023. https://doi.org/10.1111/jvs.12303

Maxwell, P. S., Eklöf, J. S., Katwijk, M. M. van, O'Brien, K. R., Torre-Castro, M. de la, Boström, C., Bouma, T. J., Krause-Jensen, D., Unsworth, R. K. F., Tussenbroek, B. I. van, & Heide, T. van der. (2017). The fundamental role of ecological feedback mechanisms for the adaptive management of seagrass ecosystems – a review. *Biological Reviews*, *92*(3), 1521–1538. https://doi.org/10.1111/brv.12294

McCullough, D. G., Werner, R. A., & Neumann, D. (1998). Fire and insects in northern and boreal forest ecosystems of North America. Annual Review of Entomology, 43, 107–127. https://doi.org/10.1146/annurev.ento.43.1.107

McFarlane, B. L., Stumpf-Allen, R. C. G., & Watson, D. O. (2006). Public perceptions of natural disturbance in Canada's national parks: The case of the mountain pine beetle (Dendroctonus ponderosae Hopkins). Biological Conservation, 130(3), 340–348. https://doi.org/10.1016/j.biocon.2005.12.029

McGarigal, K., & Romme, W. H. (2012). Modeling Historical Range of Variability at a Range of Scales: An Example Application. In Historical Environmental Variation in Conservation and Natural

Resource Management (pp. 128–145). John Wiley & Sons, Ltd. https://doi.org/10.1002/9781118329726.ch9

McIntosh, A. C. S., & Macdonald, S. E. (2013). Potential for lodgepole pine regeneration after mountain pine beetle attack in newly invaded Alberta stands. Forest Ecology and Management, 295, 11–19. https://doi.org/10.1016/j.foreco.2012.12.050

McCaffrey, D. R., & Hopkinson, C. (2017). Assessing Fractional Cover in the Alpine Treeline Ecotone Using the WSL Monoplotting Tool and Airborne Lidar. *Canadian Journal of Remote Sensing*, *43*(5), 504–512. https://doi.org/10.1080/07038992.2017.1384309

McCaffrey, D.R. & Hopkinson, C. (2020) Modeling watershed-scale historic change in the alpine Treeline ecotone using random Forest. *Canadian Journal of Remote Sensing*, 46(6), 715–732. https://doi.org/10.1080/07038992.2020.1865792

McClaran, M.P., Browning, D.M. & Huang, C. (2010) Temporal dynamics and spatial variability in desert grassland vegetation. In: Webb, R.H., Boyer, D.E. & Turner, R.M. (Eds.) *Repeat photography: methods and applications in the natural sciences*. Washington, DC: Island Press, p. 145.

McCullough, D. G., Werner, R. A., & Neumann, D. (1998). Fire and insects in northern and boreal forest ecosystems of North America. *Annual Review of Entomology*, *43*, 107–127. https://doi.org/10.1146/annurev.ento.43.1.107

McGarigal K., Cushman S.A., Ene E. (2012). FRAGSTATS v4: spatial pattern analysis program for categorical and continuous maps. Computer software program produced by the authors at the University of Massachusetts, Amherst. Available at the following web site: http://www.umass.edu/landeco/research/fragstats/fragstats.html

McIntosh, A. C. S., & Macdonald, S. E. (2013). Potential for lodgepole pine regeneration after mountain pine beetle attack in newly invaded Alberta stands. *Forest Ecology and Management*, *295*, 11–19. https://doi.org/10.1016/j.foreco.2012.12.050

Meyer, M. D., Long, J. W., & Safford, H. D. (2021). Postfire restoration framework for national forests in California. *Gen. Tech. Rep. PSW-GTR-270. Albany, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station. 204 p.*, *270*. http://www.fs.usda.gov/treesearch/pubs/61909

Millar, C.I., Stephenson, N.L., Stephens, S.L. (2007). Climate change and forests of the future: managing in the face of uncertainty. Ecological Applications. 17: 2145–2151.

Mitton, J. B., & Ferrenberg, S. M. (2012). Mountain Pine Beetle Develops an Unprecedented Summer Generation in Response to Climate Warming. *The American Naturalist*, *179*(5), E163–E171. https://doi.org/10.1086/665007

Morgan, J. L., Gergel, S. E., & Coops, N. C. (2010). Aerial Photography: A Rapidly Evolving Tool for Ecological Management. *BioScience*, *60*(1), 47–59. https://doi.org/10.1525/bio.2010.60.1.9

Morgan, P., Aplet, G. H., Haufler, J. B., Humphries, H. C., Moore, M. M., & Wilson, W. D. (1994). Historical Range of Variability: A useful tool for evaluating ecosystem change. *Journal of Sustainable Forestry*, *2*(1–2), 87–111. https://doi.org/10.1300/J091v02n01_04

Murphy, P.J. 2007. “Following the base of the foothills”: Tracing the boundaries of Jasper Park and its adjacent Rock Mountains Forest Reserve. In: MacLaren I.S. (ed.). *Culturing wilderness in Jasper National Park*. The University of Alberta Press, Edmonton, AB, Canada. pp. 71–121.

Murphy, P. J. 2007. Homesteading in the Athabasca Valley to 1910. In MacLaren, I. S., Payne, M., Murphy, P. J., Reichwein, P., McDermott, L. et al. Culturing Wilderness in Jasper National Park. The University of Alberta Press.

Nagy-Reis, M., Dickie, M., Calvert, A. M., Hebblewhite, M., Hervieux, D., Seip, D. R., Gilbert, S. L., Venter, O., DeMars, C., Boutin, S., & Serrouya, R. (2021). Habitat loss accelerates for the endangered woodland caribou in western Canada. Conservation Science and Practice, 3(7), e437. https://doi.org/10.1111/csp2.437

Natural Resources Canada. (2021) High Resolution Digital Elevation Model (HRDEM)—CanElevation Series—Product specifications. https://open.canada.ca/data/en/dataset/0fe65119-e96e-4a57-8bfe-9d9245fba06b

Nelson, G. C., Bennett, E., Berhe, A. A., Cassman, K., DeFries, R., Dietz, T., Dobermann, A., Dobson, A., Janetos, A., Levy, M., Marco, D., Nakicenovic, N., O’Neill, B., Norgaard, R., Petschel-Held, G., Ojima, D., Pingali, P., Watson, R., & Zurek, M. (2006). Anthropogenic Drivers of Ecosystem Change: An Overview. *Ecology and Society*, *11*(2). https://www.jstor.org/stable/26266018

Newman, E. A. (2019). Disturbance Ecology in the Anthropocene. Frontiers in Ecology and Evolution, 7, 147. https://doi.org/10.3389/fevo.2019.00147

Nowacki, G. J., MacCleery, D. W., & Lake, F. K. (2012). Native Americans, Ecosystem Development, and Historical Range of Variation. In Historical Environmental Variation in Conservation and Natural Resource Management (pp. 76–91). John Wiley & Sons, Ltd. https://doi.org/10.1002/9781118329726.ch6

Okamoto, R., Ide, R., & Oguma, H. (2024). Automatically drawing vegetation classification maps using digital time-lapse cameras in alpine ecosystems. *Remote Sensing in Ecology and Conservation*, *10*, 188–202. https://doi.org/10.1002/rse2.364

Paine, R. T., Tegner, M. J., & Johnson, E. A. (1998). Compounded Perturbations Yield Ecological Surprises. *Ecosystems*, *1*(6), 535–545. https://doi.org/10.1007/s100219900049

Pal, M. & Mather, P.M. (2005) Support vector machines for classification in remote sensing. *International Journal of Remote Sensing*, 26(5), 1007–1011. https://doi.org/10.1080/01431160512331314083

Parks Canada. (1997). State of the Parks 1997 Report. Retrieved March 13, 2025, from https://epe.lac-bac.gc.ca/100/202/301/state_of_the_parks/html/1997/toc.html

Parks Canada. (2008). Principles and Guidelines for Ecological Restoration in Canada's Protected Natural Areas. National Parks Directorate. Gatineau, Quebec.

Parks Canada. (2021). Ecological integrity. Retrieved March 29, 2025, from https://parks.canada.ca/nature/science/conservation/ie-ei

Parks Canada. (2023a). Jasper National Park Annual Report 2023. Retrieved February 15, 2025, from https://parks.canada.ca/pn-np/ab/jasper/gestion-management/plan/2023

Parks Canada. (2023b). Guiding Principles and Operational Policies. Retrieved March 31, 2025, from https://parks.canada.ca/agence-agency/bib-lib/politiques-policies/gestion-management/princip/sec2/part2a/part2a5

Parks Canada. (2024a). Ecological integrity of national parks. Retrieved March 28, 2025, from https://www.canada.ca/en/environment-climate-change/services/environmental-indicators/ecological-integrity-national-parks.html

Parks Canada. (2024b). Wildfire status – Jasper Wildfire Complex. Retrieved November 1, 2024, from https://parks.canada.ca/pn-np/ab/jasper/visit/feu-alert-fire/feudeforet-wildfire

Parks Canada. (2024) Indigenous Connections. Available at https://parks.canada.ca/pn-np/ab/jasper/autochtones-indigenous Accessed on December 23, 2024.

Parks Canada. (2025). The 2024 Jasper Wildfire: Frequently asked questions (FAQ) about the incident, response and recovery. Retrieved February 24, 2025, fromhttps://parks.canada.ca/pn-np/ab/jasper/visit/feu-alert-fire/feudeforet-jasper-wildfire/questions

Parks, S. A., & Abatzoglou, J. T. (2020). Warmer and Drier Fire Seasons Contribute to Increases in Area Burned at High Severity in Western US Forests From 1985 to 2017. *Geophysical Research Letters*, *47*(22), e2020GL089858. https://doi.org/10.1029/2020GL089858

Parks, S. A., Dobrowski, S. Z., Shaw, J. D., & Miller, C. (2019). Living on the edge: Trailing edge forests at risk of fire-facilitated conversion to non-forest. *Ecosphere*, *10*(3), e02651. https://doi.org/10.1002/ecs2.2651

Parks, S. A., Holsinger, L. M., Panunto, M. H., Jolly, W. M., Dobrowski, S. Z., & Dillon, G. K. (2018). High-severity fire: Evaluating its key drivers and mapping its probability across western US forests. Environmental Research Letters, 13(4), 044037. https://doi.org/10.1088/1748-9326/aab791

Parks, S. A., & Abatzoglou, J. T. (2020). Warmer and Drier Fire Seasons Contribute to Increases in Area Burned at High Severity in Western US Forests From 1985 to 2017. Geophysical Research Letters, 47(22), e2020GL089858. https://doi.org/10.1029/2020GL089858

Parks, S. A., Guiterman, C. H., Margolis, E. Q., Lonergan, M., Whitman, E., Abatzoglou, J. T., Falk, D. A., Johnston, J. D., Daniels, L. D., Lafon, C. W., Loehman, R. A., Kipfmueller, K. F., Naficy, C. E., Parisien, M.-A., Portier, J., Stambaugh, M. C., Williams, A. P., Wion, A. P., & Yocom, L. L. (2025). A fire deficit persists across diverse North American forests despite recent increases in area burned. Nature Communications, 16(1), 1493. https://doi.org/10.1038/s41467-025-56333-8

Pérez-Invernón, F. J., Gordillo-Vázquez, F. J., Huntrieser, H., & Jöckel, P. (2023). Variation of lightning-ignited wildfire patterns under climate change. *Nature Communications*, *14*(1), 739. https://doi.org/10.1038/s41467-023-36500-5

Peterson, A.T., Berthiaume, K., Klett, M. & Munroe, J.S. (2022) Linking repeat photography and remote sensing to assess treeline rise with climate warming: Mount of the Holy Cross, Colorado. *Arctic, Antarctic, and Alpine Research*, 54(1), 478–487. Available from: https://doi.org/10.1080/15230430.2022.2121245

Peterson, G., Allen, C. R., & Holling, C. S. (1998). Original Articles: Ecological Resilience, Biodiversity, and Scale. *Ecosystems*, *1*(1), 6–18. https://doi.org/10.1007/s100219900002
Peterson, G. D., Cumming, G. S., & Carpenter, S. R. (2003). Scenario Planning: A Tool for Conservation in an Uncertain World. *Conservation Biology*, *17*(2), 358–366. https://doi.org/10.1046/j.1523-1739.2003.01491.x

Peterson, G. D., Cumming, G. S., & Carpenter, S. R. (2003). Scenario Planning: A Tool for Conservation in an Uncertain World. Conservation Biology, 17(2), 358–366. https://doi.org/10.1046/j.1523-1739.2003.01491.x

Price, D. T., Alfaro, R. I., Brown, K. J., Flannigan, M. D., Fleming, R. A., Hogg, E. H., Girardin, M. P., Lakusta, T., Johnston, M., McKenney, D. W., Pedlar, J. H., Stratton, T., Sturrock, R. N., Thompson, I. D., Trofymow, J. A., & Venier, L. A. (2013). Anticipating the consequences of climate change for Canada's boreal forest ecosystems. Environmental Reviews, 21(4), 322–365. https://doi.org/10.1139/er-2013-0042

Pureswaran, D. S., Roques, A., & Battisti, A. (2018). Forest Insects and Climate Change. *Current Forestry Reports*, *4*(2), 35–50. https://doi.org/10.1007/s40725-018-0075-6

Raffa, K. F., Aukema, B. H., Bentz, B. J., Carroll, A. L., Hicke, J. A., Turner, M. G., & Romme, W. H. (2008). Cross-scale Drivers of Natural Disturbances Prone to Anthropogenic Amplification: The Dynamics of Bark Beetle Eruptions. *BioScience*, *58*(6), 501–517. https://doi.org/10.1641/B580607

Railstate (2024) Impact of Jasper Wildfire felt across CN Network. Retrieved February 15, 2025, from https://www.railstate.com/impact-of-jasper-wildfire-felt-across-cn-network/

Rhemtulla, J. M., Hall, R. J., Higgs, E. S., & Macdonald, S. E. (2002). Eighty years of change: Vegetation in the montane ecoregion of Jasper National Park, Alberta, Canada. *Canadian Journal of Forest Research*, *32*(11), 2010–2021. https://doi.org/10.1139/x02-112

Robinson A. (2015). Spread and impact of an eruptive herbivore in a novel habitat: consequences of climate change-induced range expansion [University of British Columbia]. https://doi.org/10.14288/1.0220505

Rogeau, M.-P., Parisien, M.-A., & Flannigan, M. D. (2016). Fire History Sampling Strategy of Fire Intervals Associated with Mixed- to Full-Severity Fires in Southern Alberta, Canada. Forest Science, 62(6), 613–622. https://doi.org/10.5849/forsci.15-053

Rogers, G. F., Malde, H. E., & Turner, R. M. (1984). *Bibliography of Repeat Photography Evaluating Landscape Change*. University of Utah Press.

Romme, W. H., Wiens, J., & Safford, H. D. (2012). Setting the stage: Theoretical and conceptual background of historical range of variation. *Historical Environmental Variation in Conservation and Natural Resource Management*, 3–18.

Rose, S. (2020). An evaluation of deep learning semantic segmentation for land cover classification of oblique ground-based photography (Unpublished master's thesis). *University of Victoria, Victoria*.

Roush, W., Munroe, J. S., & Fagre, D. B. (2007). Development of a spatial analysis method using ground-based repeat photography to detect changes in the alpine treeline ecotone, Glacier National Park, Montana, U.S.A. *Arctic, Antarctic, and Alpine Research*, *39*(2), 12. https://doi.org/10.1657/1523-0430(2007)39[297:DOASAM]2.0.CO;2

Safford, H. D., North, M., & Meyer, M. D. (2012a). Chapter 3: Climate change and the relevance of historical forest conditions. In: North, Malcolm, Ed. 2012. Managing Sierra Nevada Forests. Gen. Tech. Rep. PSW-GTR-237. Albany, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station. Pp. 23-45. https://research.fs.usda.gov/treesearch/41082

Safford, H. D., Wiens, J. A., Hayward, & D, G. (2012b). The Growing Importance of the Past in Managing Ecosystems of the Future. In Historical Environmental Variation in Conservation and Natural Resource Management (pp. 319–327). John Wiley & Sons, Ltd. https://doi.org/10.1002/9781118329726.ch24

Safranyik L., Carroll A.L. (2006). The biology and epidemiology of the mountain pine beetle in lodgepole pine forests. The mountain pine beetle: a synthesis of biology, management and impacts on lodgepole pine, 3–66. https://nofc.cfs.nrcan.gc.ca/publications?id=26039

Sambaraju, K. R., & Goodsman, D. W. (2021). Mountain pine beetle: An example of a climate-driven eruptive insect impacting conifer forest ecosystems. CABI Reviews, 2021. https://doi.org/10.1079/PAVSNNR202116018

Sanseverino, M. E., Whitney, M. J., & Higgs, E. S. (2016). Exploring Landscape Change in Mountain Environments With the Mountain Legacy Online Image Analysis Toolkit. *Mountain Research and Development*, *36*(4), 407–416. https://doi.org/10.1659/MRD-JOURNAL-D-16-00038.1

Scheffer, M., Carpenter, S., Foley, J. A., Folke, C., & Walker, B. (2001). Catastrophic shifts in ecosystems. *Nature*, *413*(6856), Article 6856. https://doi.org/10.1038/35098000

Seidl, R., Honkaniemi, J., Aakala, T., Aleinikov, A., Angelstam, P., Bouchard, M., Boulanger, Y., Burton, P. J., De Grandpré, L., Gauthier, S., Hansen, W. D., Jepsen, J. U., Jõgiste, K., Kneeshaw, D. D., Kuuluvainen, T., Lisitsyna, O., Makoto, K., Mori, A. S., Pureswaran, D. S., … Senf, C. (2020). Globally consistent climate sensitivity of natural disturbances across boreal and temperate forest ecosystems. *Ecography*, *43*(7), 967–978. https://doi.org/10.1111/ecog.04995

Seidl, R., Spies, T. A., Peterson, D. L., Stephens, S. L., & Hicke, J. A. (2016). Searching for resilience: Addressing the impacts of changing disturbance regimes on forest ecosystem services. *The Journal of Applied Ecology*, *53*(1), 120–129. https://doi.org/10.1111/1365-2664.12511

Seidl, R., Thom, D., Kautz, M., Martin-Benito, D., Peltoniemi, M., Vacchiano, G., Wild, J., Ascoli, D., Petr, M., Honkaniemi, J., Lexer, M. J., Trotsiuk, V., Mairota, P., Svoboda, M., Fabrika, M., Nagel, T. A., & Reyer, C. P. O. (2017). Forest disturbances under climate change. *Nature Climate Change*, *7*(6), 395–402. https://doi.org/10.1038/nclimate3303

Seidl, R., & Turner, M. G. (2022). Post-disturbance reorganization of forest ecosystems in a changing world. *Proceedings of the National Academy of Sciences*, *119*(28), e2202190119. https://doi.org/10.1073/pnas.2202190119

Simard, M., Romme, W. H., Griffin, J. M., & Turner, M. G. (2011). Do mountain pine beetle outbreaks change the probability of active crown fire in lodgepole pine forests? Ecological Monographs, 81(1), 3–24. https://doi.org/10.1890/10-1176.1

Skretting, T. 2024. Impacts of mountain pine beetle outbreak and wildland fuel reduction in Jasper National Park, Alberta, Canada. [Unpublished masters thesis] University of Lethbridge.

Skretting, T., Chasmer, L., Watson, C., James, P. M. A., Townshend, I., & Perrakis, D. (2025). Forest fuel structure and loading along a gradient of gray-phase mountain pine beetle severity in Jasper National Park, Alberta, Canada. Canadian Journal of Forest Research. https://doi.org/10.1139/cjfr-2024-0319

Standish, R. J., Thompson, A., Higgs, E. S., & Murphy, S. D. (2013). Concerns about Novel Ecosystems. In Novel Ecosystems (pp. 296–309). John Wiley & Sons, Ltd. https://doi.org/10.1002/9781118354186.ch37

Steffen, W., Broadgate, W., Deutsch, L., Gaffney, O. & Ludwig, C. (2015) The trajectory of the Anthropocene: the great acceleration. *The Anthropocene Review*, 2(1), 81–98. https://doi.org/10.1177/2053019614564785

Stephenson, N.L., Millar, C.I., Cole, D.N. (2010). Shifting environmental foundations: the unprecedented and unpredictable future. In: Cole, D.N., Yung, L., eds. Beyond naturalness: rethinking park and wilderness stewardship in an era of rapid change. Washington, DC: Island Press: 50–66.

Stevens-Rumann, C. S., Kemp, K. B., Higuera, P. E., Harvey, B. J., Rother, M. T., Donato, D. C., Morgan, P., & Veblen, T. T. (2018). Evidence for declining forest resilience to wildfires under climate change. Ecology Letters, 21(2), 243–252. https://doi.org/10.1111/ele.12889

Stockdale, C. A., Bozzini, C., Macdonald, S. E., & Higgs, E. (2015). Extracting ecological information from oblique angle terrestrial landscape photographs: Performance evaluation of the WSL Monoplotting Tool. *Applied Geography*, *63*, 315–325. https://doi.org/10.1016/j.apgeog.2015.07.012

Stralberg, D., Wang, X., Parisien, M.-A., Robinne, F.-N., Sólymos, P., Mahon, C. L., Nielsen, S. E., & Bayne, E. M. (2018). Wildfire-mediated vegetation change in boreal forests of Alberta, Canada. Ecosphere, 9(3), e02156. https://doi.org/10.1002/ecs2.2156

Swetnam, T. W., Allen, C. D., & Betancourt, J. L. (1999). Applied Historical Ecology: Using the Past to Manage for the Future. *Ecological Applications*, *9*(4), 1189–1206. https://doi.org/10.1890/1051-0761(1999)009[1189:AHEUTP]2.0.CO;2

Tande, G. F. (1979). Fire history and vegetation pattern of coniferous forests in Jasper National Park, Albert. *Canadian Journal of Botany*, *57*(18), 1912–1931. https://doi.org/10.1139/b79-241

Tande, Gerald F. 1980. Interpreting fire history in Jasper National Park, Alberta. Pages 31-34 in: Stokes, Marvin A.; Dieterich, John H. (editors), Proceedings of the Fire History Workshop: October 20-24, 1980, Tucson, Arizona. General Technical Report RM-GTR-81. Ft. Collins, CO: US Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station. 142 p.

Taylor, S. W., & Carroll, A. L. 2003. Disturbance, forest age, and mountain pine beetle outbreak dynamics in BC: A historical perspective. In *Mountain pine beetle symposium: Challenges and solutions* (Vol. 3031). Natural Resources Canada, Canadian Forest Service, Pacific Forestry Centre Victoria.

Tedim, F., Leone, V., Amraoui, M., Bouillon, C., Coughlan, M. R., Delogu, G. M., Fernandes, P. M., Ferreira, C., McCaffrey, S., McGee, T. K., Parente, J., Paton, D., Pereira, M. G., Ribeiro, L. M., Viegas, D. X., & Xanthopoulos, G. (2018). Defining Extreme Wildfire Events: Difficulties, Challenges, and Impacts. Fire, 1(1), Article 1. https://doi.org/10.3390/fire1010009

Trant, A., Higgs, E. & Starzomski, B.M. (2020) A century of high elevation ecosystem change in the Canadian Rocky Mountains. *Scientific Reports*, 10(1), 9698. https://doi.org/10.1038/s41598-020-66277-2

Trant, A.J., Starzomski, B.M. & Higgs, E. (2015) A publically available database for studying ecological change in mountain ecosystems. *Frontiers in Ecology and the Environment*, 13(4), 187. https://doi.org/10.1890/15.WB.007

Tricker, J., Lantz, T. C., Rhemtulla, J. M., & Higgs, E. S. (2025). A tale of two disturbances: Can mountain pine beetle restore landcover composition and pattern altered by fire suppression in Jasper National Park? Regional Environmental Change, 25(1), 21. https://doi.org/10.1007/s10113-024-02356-8

Tricker, J., Wright, C., Rose, S., Rhemtulla, J., Lantz, T., & Higgs, E. (2024). Assessing the accuracy of georeferenced landcover data derived from oblique imagery using machine learning. *Remote Sensing in Ecology and Conservation*, *10*(3), 401–415. https://doi.org/10.1002/rse2.379

Trzcinski, M. K., & Reid, M. L. (2008). Effect of management on the spatial spread of mountain pine beetle (Dendroctonus ponderosae) in Banff National Park. Forest Ecology and Management, 256(6), 1418–1426. https://doi.org/10.1016/j.foreco.2008.07.003

Tulloch, A., Possingham, H. P., & Wilson, K. (2011). Wise selection of an indicator for monitoring the success of management actions. Biological Conservation, 144(1), 141–154. https://doi.org/10.1016/j.biocon.2010.08.009

Turner N. (1999). Time to burn:" traditional use of fire to enhance resource production by Aboriginal peoples in British Columbia. In: Indians F (ed) and the Land in the Pacific Northwest by Robert Boyd. Oregon State University Press, Corvallis OR, pp 158–218

Turner, M. G. (2010). Disturbance and landscape dynamics in a changing world. *Ecology*, *91*(10), 2833–2849. https://doi.org/10.1890/10-0097.1

Turner, M. G., Braziunas, K. H., Hansen, W. D., Hoecker, T. J., Rammer, W., Ratajczak, Z., Westerling, A. L., & Seidl, R. (2022). The magnitude, direction, and tempo of forest change in Greater Yellowstone in a warmer world with more fire. *Ecological Monographs*, *92*(1), e01485. https://doi.org/10.1002/ecm.1485

Turner, M. G., Donato, D. C., & Romme, W. H. (2013). Consequences of spatial heterogeneity for ecosystem services in changing forest landscapes: Priorities for future research. Landscape Ecology, 28(6), 1081–1097. https://doi.org/10.1007/s10980-012-9741-4

Turner, M. G., & Seidl, R. (2023). Novel Disturbance Regimes and Ecological Responses. *Annual Review of Ecology, Evolution, and Systematics*, *54*(1), null. https://doi.org/10.1146/annurev-ecolsys-110421-101120

Viljur, M.-L., Abella, S. R., Adámek, M., Alencar, J. B. R., Barber, N. A., Beudert, B., Burkle, L. A., Cagnolo, L., Campos, B. R., Chao, A., Chergui, B., Choi, C.-Y., Cleary, D. F. R., Davis, T. S., Dechnik-Vázquez, Y. A., Downing, W. M., Fuentes-Ramirez, A., Gandhi, K. J. K., Gehring, C., … Thorn, S. (2022). The effect of natural disturbances on forest biodiversity: An ecological synthesis. Biological Reviews, 97(5), 1930–1947. https://doi.org/10.1111/brv.12876

Visser, M. E. (2008). Keeping up with a warming world; assessing the rate of adaptation to climate change. Proceedings of the Royal Society B: Biological Sciences, 275(1635), 649–659. https://doi.org/10.1098/rspb.2007.0997

Walter, T. (2020) Satellite-based augmentation systems (SBASs). In: Morton, Y.J., van Diggelen, F., Spilker, J.J., Jr., Parkinson, B.W., Lo, S. & Gao, G. (Eds.) *Position, navigation, and timing technologies in the 21st century.* Hoboken, NJ: John Wiley & Sons, Ltd., pp. 277–306. https://doi.org/10.1002/9781119458449.ch13

Wasserman, T. N., & Mueller, S. E. (2023). Climate influences on future fire severity: A synthesis of climate-fire interactions and impacts on fire regimes, high-severity fire, and forests in the western United States. *Fire Ecology*, *19*(1), 43. https://doi.org/10.1186/s42408-023-00200-8

Webb, R. H., Boyer, D. E., & Turner, R. M. (Eds.). (2010). *Repeat Photography: Methods and Applications in the Natural Sciences* (1 edition). Island Press.

Weed, A. S., Ayres, M. P., & Hicke, J. A. (2013). Consequences of climate change for biotic disturbances in North American forests. *Ecological Monographs*, *83*(4), 441–470. https://doi.org/10.1890/13-0160.1

White P.S., Pickett S.T.A. (1985). Chapter 1 - Natural disturbance and patch dynamics: an introduction. In Pickett STA, White PS (Eds) The ecology of natural disturbance and patch dynamics 3–13 Academic Press. https://doi.org/10.1016/B978-0-12-554520-4.50006-X

White, R. H., & Zipperer, W. C. (2010). Testing and classification of individual plants for fire behaviour: Plant selection for the wildland–urban interface. International Journal of Wildland Fire, 19(2), 213–227. https://doi.org/10.1071/WF07128

Whitman, E., Parisien, M.-A., Thompson, D. K., & Flannigan, M. D. (2019). Short-interval wildfire and drought overwhelm boreal forest resilience. Scientific Reports, 9(1), 18796. https://doi.org/10.1038/s41598-019-55036-7

Whitman, E., Parks, S. A., Holsinger, L. M., & Parisien, M.-A. (2022). Climate-induced fire regime amplification in Alberta, Canada. *Environmental Research Letters*, *17*(5), 055003. https://doi.org/10.1088/1748-9326/ac60d6

Woodley, S. (1995). Playing with fire: vegetation management in the Canadian Parks Service. *General technical report INT (USA)*, (320).

Wotton, B. M., Flannigan, M. D., & Marshall, G. A. (2017). Potential climate change impacts on fire intensity and key wildfire suppression thresholds in Canada. Environmental Research Letters, 12(9), 095003. https://doi.org/10.1088/1748-9326/aa7e6e

Wright, C., Bone, C., Mathews, D., Tricker, J., Wright, B., & Higgs, E. (2024). Mountain Image Analysis Suite (MIAS): A new plugin for converting oblique images to landcover maps in QGIS. *Transactions in GIS*, *28*(7), 2044–2061. https://doi.org/10.1111/tgis.13229

Wright H.E., Heinselman M.L. (1973). The ecological role of fire in natural conifer forests of western and northern North America. Quaternary Research 3(3):317–318. https://doi.org/10.1016/0033-5894(73)90001-X

Wulder, M. (1998) Optical remote-sensing techniques for the assessment of forest inventory and biophysical parameters. *Progress in Physical Geography*, 22(4), 449–476. https://doi.org/10.1177/030913339802200402

Wulder, M.A., White, J.C., Hay, G.J. & Castilla, G. (2008) Towards automated segmentation of forest inventory polygons on high spatial resolution satellite imagery. *The Forestry Chronicle*, 84(2), 221–230. https://cfs.nrcan.gc.ca/publications?id=29414

Wurtzebach, Z., & Schultz, C. (2016). Measuring Ecological Integrity: History, Practical Applications, and Research Opportunities. BioScience, 66(6), 446–457. https://doi.org/10.1093/biosci/biw037

Zou, Y., Yu, Z., Kumar, B.V.K.V. & Wang, J. (2018) Domain adaptation for semantic segmentation via class-balanced self-training (arXiv:1810.07911). *arXiv*. https://doi.org/10.48550/arXiv.1810.07911

Appendix A: Image list

List of MLP oblique photographs used for Jasper National Park study area.

rse2379-sup-0001-Appendix_A.pdf

Appendix B: Images and classifications

Oblique images and associated landcover classifications.

rse2379-sup-0001-Appendix_A.pdf

Appendix C: Georeferenced landcover classifications

19 georeferenced landcover grids.

rse2379-sup-0002-Appendix_C.zip

Appendix D: Maps

Oblique and orthogonal landcover maps.

rse2379-sup-0003-Appendix_D.zip

Appendix E: Image list

List of historical and repeat images and associated metadata.

Station name	Historical image #	Repeat image #	Repeat year	Image azimuth	Latitude	Longitude	Weblink
Cairgorm I	e011313366	DSCF4304	2022	140.85	52.93066	-118.19186	https://explore.mountainlegacy.ca/historic_captures/show/23792
Clairvaux I	e011214103	DSCF4444	2022	321.6	52.84759	-118.38385	https://explore.mountainlegacy.ca/historic_captures/show/23735
	e011214105	DSCF4447		10.6	52.84759	-118.38385	https://explore.mountainlegacy.ca/historic_captures/show/23737
Hawk Mtn	e006609622	DSCF0614	2020	190	53.01567	-118.01779	https://explore.mountainlegacy.ca/historic_captures/show/23657
	e006609623	DSCF0617		224.75	53.01567	-118.01779	https://explore.mountainlegacy.ca/historic_captures/show/23658
	e006609624	DSCF0623		259.75	53.01567	-118.01779	https://explore.mountainlegacy.ca/historic_captures/show/23659
Morro Peak I	e011313299	DSCF0575	2020	273.7	53.03387	-118.06708	https://explore.mountainlegacy.ca/historic_captures/show/23592
Mt Elysium	e011305802	DSCF4106	2022	212	52.95276	-118.35934	https://explore.mountainlegacy.ca/historic_captures/show/23524
	e011305801	DSCF4127		182.6	52.95263	-118.35904	https://explore.mountainlegacy.ca/historic_captures/show/23523
Mt Esplanade	e011313396	DSCF0660	2020	100	53.07630	-118.15224	https://explore.mountainlegacy.ca/historic_captures/show/23507
Mt Greenock	e011313398	DSCF0547	2020	185.75	53.10518	-118.08796	https://explore.mountainlegacy.ca/historic_captures/show/23492
Mt Henry	e011214151	DSCF4336	2022	160.1	52.93344	-118.25096	https://explore.mountainlegacy.ca/historic_captures/show/23490
	e011214152	DSCF4338		205.6	52.93344	-118.25096	https://explore.mountainlegacy.ca/historic_captures/show/23491
Muhegan Mtn I	e011214115	DSCF4247	2022	301.6	52.83339	-118.23790	https://explore.mountainlegacy.ca/historic_captures/show/23430
	e006609619	DSCF4236		329.1	52.83343	-118.23745	https://explore.mountainlegacy.ca/historic_captures/show/23428
	e011214113	DSCF4238		3.1	52.83343	-118.23745	https://explore.mountainlegacy.ca/historic_captures/show/23427
	e011214112	DSCF4239		37.6	52.83343	-118.23745	https://explore.mountainlegacy.ca/historic_captures/show/23426
Pyramid Mtn	e011214051	DSCF0782	2020	102.25	52.95138	-118.14895	https://explore.mountainlegacy.ca/historic_captures/show/23402
	e011214055	DSCF0770		154	52.95107	-118.14916	https://explore.mountainlegacy.ca/historic_captures/show/23406
	e002506792	DSCF0768		124.75	52.95107	-118.14916	https://explore.mountainlegacy.ca/historic_captures/show/37462
Roche Bonhomme I	e011313312	DSCF0716	2020	230.75	52.94677	-117.96340	https://explore.mountainlegacy.ca/historic_captures/show/23375
	e006609620	DSCF0725		287	52.94688	-117.96308	https://explore.mountainlegacy.ca/historic_captures/show/23384
Signal Mtn	e011313274	DSCF4015	2022	272.6	52.86455	-117.97898	https://explore.mountainlegacy.ca/historic_captures/show/23356
The Palisade	e006609627	DSCF0803	2020	75.75	52.99340	-118.13670	https://explore.mountainlegacy.ca/historic_captures/show/23326
	e011313409	DSCF0838		27.25	52.99345	-118.13670	https://explore.mountainlegacy.ca/historic_captures/show/23325
The Whistlers II	e011313287	DSCF1188	2022	39.6	52.83005	-118.12465	https://explore.mountainlegacy.ca/historic_captures/show/23317

Appendix F: Image classifications

The classifications of the oblique images are available in the Borealis MountainScape Segmentation Dataset:

https://borealisdata.ca/dataset.xhtml?persistentId=doi:10.5683/SP3/CEYU10.

Appendix G: Confusion matrix

Confusion matrix for the 2020/22 landcover map compared to reference aerial imagery using 400 equalized stratified random points.

		Reference data									
	Landcover class	Mature conifer	Barren ground	Water	Shrub	Wetland	Broadleaf/ mixedwood	Regenerating conifer	Herbaceous	Total	Users Accuracy
Classified data	Mature conifer	40	0	0	2	1	7	0	0	50	0.8
	Barren ground	6	32	4	5	0	1	1	1	50	0.64
	Water	1	1	47	0	1	0	0	0	50	0.94
	Shrub	3	5	3	37	0	1	0	1	50	0.74
	Wetland	3	0	6	2	39	0	0	0	50	0.78
	Broadleaf/ mixedwood	9	0	0	2	2	37	0	0	50	0.74
	Regenerating conifer	0	0	0	13	0	0	37	0	50	0.74
	Herbaceous	0	1	0	2	0	0	0	47	50	0.94
	Total	62	39	60	63	43	46	38	49	400	0
	Producers Accuracy	0.65	0.82	0.78	0.59	0.91	0.8	0.97	0.96	0	Overall accuracy = 0.79 (Kappa = 0.76)