TOWARDS IMPROVED UNDERSTANDING AND MANAGEMENT OF MIXED-SEVERITY FIRE REGIMES IN MOUNTAIN FORESTS

by

Hélène Marcoux

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Abstract

Understanding spatial and temporal patterns of fire regimes is critically important for sustainable forest management and fire hazard mitigation. Mixed-severity fire regimes, in particular, are poorly understood, yet increasingly recognized as important drivers of stand- and landscape-heterogeneity. I address knowledge gaps pertaining to the management and understanding of mixed-severity regimes including: (1) classification and mapping, (2) prevalence in mountain forests, (3) underlying topographical drivers, and (4) stand dynamics. Research questions were addressed using dendrochronological field data (fire scars, tree establishment dates, stand composition and structure) from 20 randomly selected research sites in southeastern British Columbia,

I examined whether mixed-severity regimes, as currently represented in fire-regime classification schemes, led to erroneous landscape-level fire regime mapping. I used my field data to evaluate the accuracy of two classification systems (Natural Disturbance Type (NDT) and Historical Natural Fire Regime (HNFR)) used by managers to map fire regimes in British Columbia (Chapter 2). Each classification system made considerable and contrasting errors in identifying mixed-severity regimes relative to the field data and these misrepresentations were tied to elevation. I attributed these errors to assumptions about disturbances underlying each classification system, as well as limitations of the research methods used to estimate fire frequency (i.e., using either stand-age or fire-scar data in isolation).

I explored the prevalence of mixed-severity fire regimes, importance of underlying topographic drivers, as well as the influence of mixed- *versus* high-severity fires on forest composition and structure (Chapter 3). I found evidence of mixed-severity fires at 55%. At these sites, most reconstructed fires (73%) were documented solely by fire scars, indicating many were of low-to-moderate severity. The remaining 27% of fires were severe enough to create conditions suitable for even-aged cohort to establish. Spatial patterns of fire severity were primarily controlled by elevation (i.e., severity increased with elevation). Composition varied with disturbance history; however, structural differences (e.g., tree size classes) were subtle, with the exception of snag densities, which were much greater in old, high-severity forests (where time-since-last-fire >250 years). Understanding the ecological heterogeneity created by mixed-severity regimes potentially influences decisions related to conservation, silviculture, wildfire and fuel mitigation.

Preface

This research is part of a multi-disciplinary research program investigating fire history in southeastern British Columbia. I used pre-established research sites (by Da Silva 2009), but held primary responsibility for the sample design, the collection and analysis of data, as well as the writing and submission of manuscripts. During this process, I received considerable guidance, input and support from my two supervisors, Dr. Lori Daniels and Dr. Sarah Gergel, who are both listed as co-authors of the manuscript developed from Chapter 2.

Chapter 2: Accepted May 13, 2013. Marcoux, H.M., Gergel, S.E., and Daniels, L.D. Mixed-severity fire regimes: How well are they represented by existing fire-regime classification systems? Canadian Journal of Forest Research. Reprinted with permission from NRC Research Press. Comments: This study was conducted and written by HMM under the supervision of SEG and LDD who helped conceptualize the study and contributed to the preparation of the manuscript.

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List of Abbreviations

BEC − Biogeoclimatic ecosystem classification

FRI − Fire return interval

HNFR − Historical Natural Fire Regime

JGC – Joseph and Gold Creek watersheds

NDT − Natural Disturbance Type

TSLF − Time since last fire

Acknowledgements

The first half of this MSc degree was filled with excitement and the type of curiosity that comes with a fresh research project; including a fantastic summer in the field and months of lab work dating tree-rings. I was free to immerse myself in my studies and indulge in the grad student lifestyle. The second half of my degree overlapped with two of the biggest changes in my life: the birth of my son and the death of my mother. With the support of many, I was able to finish this thesis despite these lifealtering events.

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This long journey has come to its end, but could not have been possible without the love, support and patience from my dear family and friends. To my loving partner Gavin, I am excited for a new “thesis and school free” chapter in our life. To my awesome sister, you’re the best volunteer and friend I have ever had. To my dad, thanks for encouraging me to follow my passions.

To my dear maman…

You would have been so proud (and relieved!) to see this thesis come to completion.

(I am sure you can download a copy if you have internet access in the afterlife.)

Chapter 1: Introduction

1.1 Importance of fire history research

Apparent increases in wildfire prevalence and severity in western United States and Canada in recent years have raised concerns for communities living at the wildland-urban interface (Filmon 2003; Flannigan et al. 2005; Westerling et al. 2006). Several interacting factors may be driving these large fire events: climate change, insect outbreaks, fire suppression, and land-use practices that exclude fire from the landscape (e.g., silviculture, cattle grazing). For example, increases in occurrence of severe fires in mid-elevation forests of the northern Rocky Mountains have been attributed to warming climates (Westerling et al. 2006). At low elevations, land-use that excludes low-to-moderate severity fires may have promoted unnaturally dense forests with increased fire risk (Schoennagel et al. 2004). Widespread mortality driven by mountain pine beetle outbreaks may also increase forest susceptibility to severe wildfire (Schoennagel et al. 2012).Teasing out causal factors driving contemporary fires requires understanding historic fire regimes and their underlying drivers. Site-specific information is critical if we are to determine whether present-day fires are more severe than the historic "norm" for a given ecosystem (Veblen 2003).

Wildland fires shape forested landscapes in western North America (Agee 1993; Bowman et al. 2009), making knowledge of their ecology important for sustainable forest management. Specifically, understanding forest and fire dynamics is essential for developing silvicultural practices that emulate natural disturbances to maintain biodiversity and meet conservation objectives (Landres et al. 1999; Cissel et al. 1999). Application of this management concept requires an understanding of the spatial and temporal variability of disturbance agents (Wong and Iverson 2004). In southeastern British Columbia (B.C.), this includes the historic characteristics and effects of fire regimes, which were fundamental in shaping ecosystem patterns, structures and processes.

1.2 Changing fire regime paradigms

The concept of a 'fire regime' is used to describe the spatial and temporal dimensions of fire for a given area (Agee 1993; Schoennagel et al. 2004). Fires and their impacts can be described using attributes such as location, timing (or seasonality), extent (or size), shape, frequency , intensity, duration and severity. Commonly, fire regimes are grouped by severity (low, mixed, high) or frequency (measured as return intervals; 0-35 yr, 35-100 yr, 100+ yr) (Schmidt et al. 2003; Agee 2004). Severity is defined as the impact on dominant vegetation, a function of fire characteristics (e.g., intensity and duration) and species' fire tolerance (Agee 2004). Until the last decade, most research focused on two contrasting fire regime types (Perry et al. 2011). Low-severity regimes are characterized by frequent, non-lethal surface fires that maintain stand composition and structure (e.g., "stand-maintaining") and high-severity regimes are characterized by infrequent high intensity crown fires that kill the most vegetation and replace stands (e.g., "stand-replacing")(Fig. 1.1). It is now recognized that classifying fire regimes as solely low-severity or high-severity oversimplifies variability inherent of many forest ecosystems (Perry et al. 2011).

Mixed-severity fires and regimes represent a new paradigm in fire ecology by expanding beyond simple binary classes of low and high severity. Relatively few studies have examined these more spatially and temporally heterogeneous mixed-severity regimes

(Fig. 1.1) (Agee 2004; Perry et al. 2011). As a result, their spatial extent is poorly understood. Recently, mixed severity fire regimes have been documented in forests traditionally typified as either low- (e.g., Ponderosa pine (*Pinus ponderosa*)) or high-severity systems (e.g., lodgepole pine (*Pinus contorta*) and mixed-conifer) (Klenner et al. 2008; Hessburg et al. 2007; Sherriff and Veblen 2006). The concept of ‘mixed-severity’ is applicable to individual fires and fire regimes. A mixed-severity fire has variable effects within the boundary of the fire (e.g., low, moderate and severity patches). With a mixed-severity regime, severity is patchy and variable among fire events (Agee 2004; Perry et al. 2011). Fire frequency is also highly variable and can range from short (<10 yrs) to long (>100 yrs) intervals between fires (Agee 2004). Some patch dynamics and ecological responses to fire are unique to mixed-severity systems (Halofsky et al. 2011) which may contain many small and very few large patches (Perry et al. 2011) (Fig. 1.1). Consequently, forests have a high edge to interior ratio likely resulting in a greater diversity of wildlife and vegetation than low- or high-severity systems (Perry et al. 2011).

1.3 Drivers of mixed-severity regimes

Mixed-severity fire regimes are influenced by complex interactions between top-down (e.g., weather, climate) and bottom-up (e.g., fuels, species composition) controls on fire (Perry et al. 2011). All of these forcing agents influence each fire regime type to some degree, although some are more limiting than others in different regime types, ecosystems and geographic regions. High-severity regimes are associated with mesic forests (e.g., subalpine fir - Engelmann spruce (Abies lasiocarpa – Picea engelmannii), where extreme dry climatic conditions are the limiting factor controlling fire spread rather than fuel levels (Johnson et al. 1990; Schoennagel et al. 2004). Low-severity regimes are associated with dry forests (such as low elevation ponderosa pine – Douglas fir), where factors influencing fuel levels and configuration are more limiting to fire spread than dry weather. In contrast, much of the variability within mixed-severity regimes is driven by the interaction between regional climatic and local topo-edaphic controls on fire behavior and fuel levels (Heyerdahl et al. 2001; Beaty and Taylor 2007). These same factors likely influence the proportion of low to high severity patches within a single fire and regime of a particular region (Heyerdahl et al. 2002, Gedalof et al. 2005). Similarly, mixed-severity fire regimes are thought to occur as an intermediate between low- and high-severity regimes along environmental gradients. For instance, in some landscapes mixed-severity regimes occur at mid-elevations where complex topography and diverse forest composition exist (Fig. 1.1) (Schoennagel et al. 2004; Beaty and Taylor 2008).

1.4 Research approaches

Growing evidence of mixed-severity fires stem from diverse research methods used to study contemporary and historical fires. Mixed-severity effects of fires burning since the 1980s have been documented using field measurements and aerial photographs taken before and after a fire (Thompson and Spies 2010). Remote sensing of pre- and post-fire conditions is used to quantify burn severity and identify complex fire severity patterns within fires (Soverel et al. 2010). Severity can also be inferred using forest structural attributes derived from aerial photographs coupled with stand development theory (Sherriff and Veblen 2007; Hessburg et al. 2007). This last approach found mixed-severity regimes to be common in parts of the United-States since the late 1800s (Sherriff and Veblen 2007; Hessburg et al. 2007). In contrast, quantifying historical mixed-severity regimes over the past 300-500 years requires different approaches than those used to examine contemporary fires. Historical reconstructions use dendrochronology techniques including stand-age demographics and fire-scar analyses (Taylor and Skinner 1998; 2003; Sherriff and Veblen 2006; Brown et al. 2008; Heyerdahl et al. 2012). This approach combines direct (i.e., cambial fire scars) with indirect (i.e., even-aged cohorts) lines of evidence to infer past fire severity and frequency.

Knowledge of mixed-severity regimes has improved using research approaches that combine multiple lines of evidence, rather than a single source of evidence. It is becoming clear that dendrochronological approaches must be adapted to the type of fire severity pattern observed. Traditionally, low- severity regimes have been studied using fire scars to identify fire frequency (Brown and Swetnam 1994). In contrast, after high-severity fires, scarred trees may not be present requiring other lines of evidence (e.g., even-aged cohorts of trees, fire scars) (Margolis et al. 2007). Understanding mixed-severity regimes requires a combination of methods (Sherriff and Veblen 2006), which few studies have attempted until quite recently (e.g., Taylor and Skinner 2003; Brown et al. 2008; Heyerdahl et al. 2012).

1.5 Mapping fire regimes

Maps depicting the spatial extent of historic fire regimes have become an increasingly important tool for sustainable forest management and wildfire risk assessments (Morgan et al. 2001) requiring landscape-level information about historical fires. Fire regime classification systems have been developed to map severity and frequency often using local studies or expert knowledge to extrapolate to the broader landscape. British Columbia has two fire regime classification systems, each yielding different interpretations of fire regimes for a given ecosystem (Table 1.1). These discrepancies may be particularly elevated for the less well understood mixed-severity regime type.

1.6 Research objectives

This thesis includes two studies that improve our understanding of mixed-severity fire regimes. I used detailed dendrochronological reconstructions of fire history using fire scars and tree establishment dates from 20 randomly selected sites within two study watersheds to address three primary questions. My research also contributes to a growing body of knowledge about mixed-severity regimes with a focus on mixed-conifer forests in southeastern B.C., I also address critical research gaps in dendrochronology methodologies used for estimating fire severity.

Firstly, given the poor understanding of mixed-severity regimes, I determined whether considerable errors exist in fire-regime classifications used in landscape-level mapping. In Chapter 2, I asked: *How well do existing fire-regime classification systems represent mixed-severity fire regimes?* I used stand-level data to evaluate the accuracy of two classification systems (Natural Disturbance Type (NDT) and Historical Natural Fire Regime (HNFR)) used by forest managers to map fire regimes in British Columbia, Canada. Discrepancies are attributed to natural disturbance assumptions underlying these two classification systems, as well as limitations of research methods used to estimate fire frequency (i.e., methods specific to low- *versus* high-severity fire regimes).

In Chapter 3, I explored the prevalence of mixed-severity fire regimes, importance of underlying topographic drivers, as well as the influence of these fire regimes on forest composition and structure. Here I asked: *What is the prevalence of mixed-severity fire across the landscape and how is it influenced by topography? How do fire, topography and their interaction influence forest composition and structure*? I differentiated each stand into three fire history groups where the prevalence of low-to-moderate severity fires was inferred using fire scars, while even-aged cohorts were used as indicators of high-severity fires. Time since last fire differentiated stand development pathways among the three fire history groups. Lastly, I examined the influence of topographical variables species composition and structural attributes.

In Chapter 4, I highlighted the significance of my findings within the context of other fire history research methods, and explored the management implications and potential applications of my research for mixed-conifer forests of British Columbia.

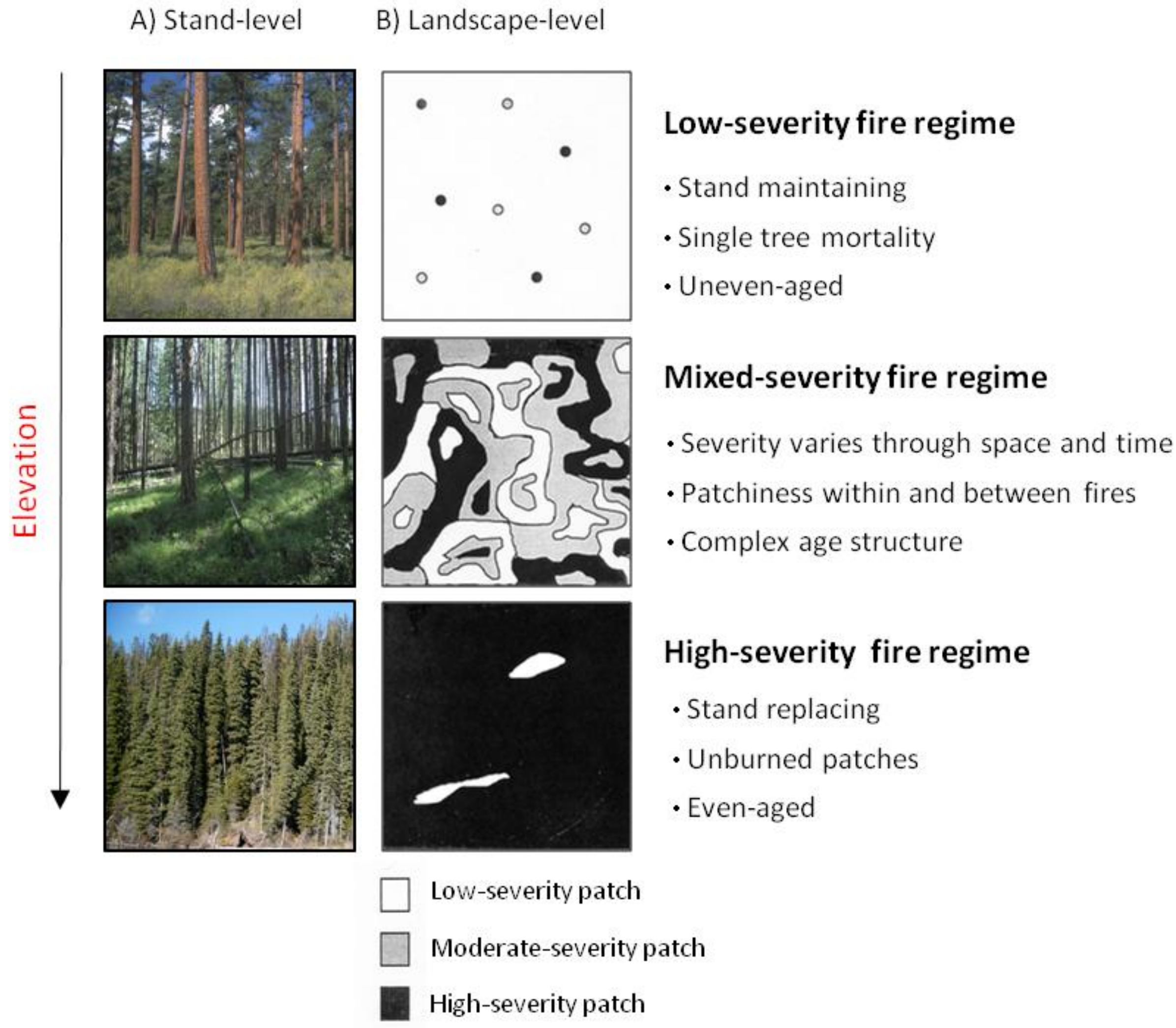

Fig. 1.1. The characteristics of each fire regime type including A) stand-level example of forest structure and B) landscape-level depictions of patch configurations from a bird's-eye view (adapted from Agee 2004).

Table 1.1 Comparison of the two fire history classification schemes, Natural Disturbance Types (NDT) and Historic Natural Fire Regime (HNFR), applied to the 10 most common forest ecosystems in southeastern British Columbia. The ecosystem types shown in bold dominate the Joseph Gold Creek watersheds examined in this study. Ecosystems are stratified by topography in the HNFR system, but not the NDT system.

Classification System	Fire Severity	Fire Return Interval (mean or range)	Topography (Aspect/Slope)		
			Level/Gentle	Warm/Steep	Cool/Steep
Natural Disturbance Type					
NDT1	High	350 yrs	ESSFwm		
NDT2	High	200 yrs	--		
NDT3	**High**	**150 yrs**	**MSdk**, ICHdw, ICHdm, ICHmk, **ESSFdk**, ESSFdkw		
NDT4	Low	4-50 yrs	IDFdm		
NDT5	Little or no occurrence of fire		ESSFdkp, IMAun		
Historic Natural Fire Regime					
HNFR0	Little or no occurrence of fire		IMAun	IMAun	IMAun
HNFR1	Low	0-35 yrs	IDFdm	IDFdm	--
HNFR2	**Mixed**	**0-35 yrs**	**MSdk,** ICHdw ICHdm, ICHmk	**MSdk,** ICHdw ICHdm, ICHmk	IDFdm
HNFR3	High	0-35 yrs	--	--	--
HNFR4	**Mixed**	**35-100 yrs**	**ESSFdk,** ESSFwm ESSFdkw	**ESSFdk,** ESSFwm ESSFdkw	**MSdk,** ICHmk ICHdw, ICHdm
HNFR5	**High**	**35-100 yrs**	**--**	**--**	**ESSFdk,** ESSFdkw
HNFR6	Mixed	100-200 yrs	--	--	ESSFwm
HNFR7	High	100-200 yrs	--	--	--
HNFR8	High	≥200 yrs	ESSFdkp	ESSFdkp	ESSFdkp

Biogeoclimatic Ecosystem Classification Unit Abbreviations:
ICHdw/dm/mk: Interior Cedar Hemlock dry warm/ dry mild/ moist cool
IDFdm: Interior Douglas-Fir dry moist
IMAun: Interior Mountain-Heather Alpine undifferentiated
MSdk: Montane Spruce dry cool
ESSFdk/wm/dkw/dkp: Engelmann Spruce-Subalpine Fir dry cool/ wet mild / dry cool woodland/ dry cool parkland

Chapter 2: Mixed-severity fire regimes: How well are they represented by existing fire-regime classification systems?[1]

2.1 Introduction

Traditionally, fire regimes have been categorized as dominated by either infrequent, severe fires or frequent, low-severity fires (Perry et al. 2011). With improved understanding of wildfire complexity in mountain forests, the concept of a "mixed-severity fire regime" has emerged as a new paradigm guiding research, management and conservation at stand- to landscape-levels (Taylor and Skinner 1998; Schoennagel et al. 2004). The mixed-severity fire regime explicitly recognizes that interactions of topography, fuels and weather in mountains lead to diverse effects within and among individual fires (Perry et al. 2011). Cumulatively, mixed-severity fire regimes result in complex landscapes comprised of stands that last burned at a range of fire severities, as well as stands that have not burned for long periods. A growing body of evidence suggests mixed-severity fire regimes were historically more common than previously documented in North America (e.g., Sherriff and Veblen 2006; Hessburg et al. 2007; Margolis and Balmat 2009) and likely in other temperate forests worldwide. Despite their likely prevalence in many forested regions, this regime type remains poorly understood (Perry et al. 2011).

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[1] A version of this chapter has been accepted for publication. Marcoux, H.M., Gergel, S.E. and Daniels, L.D. In press. Mixed-severity fire regimes: How well are they represented by existing fire-regime classification systems? Canadian Journal of Forest Research.

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Fire-regime classifications are often used to map broad, landscape-level distributions of fire classes and provide information about the spatial and temporal variability of historic fires (Morgan et al. 2001). They are foundational to management that emulates natural fire for conservation and economic targets (Landres et al. 1999; Wong and Iverson 2004). For example, fire regime classes are used to guide silviculture to retain forest structures and composition consistent with historic disturbance regimes and provide target values for the relative abundance of different age classes (e.g., B.C. Ministry of Forests 1995; Cissel et al. 1999; Bergeron et al. 2002). With respect to wildfire management, maps of fire regime classes help identify areas where fire exclusion has increased risk to property and ecosystems (Schoennagel et al. 2004). Detailed knowledge of historic fire regimes is lacking in many forest types (Morgan et al. 2001; Wong and Iverson 2004; Swetnam and Brown 2010); thus, many fire regime classification systems rely on extrapolations from research conducted at localized scales or draw from expert knowledge (see example of Fire Regime Condition Class *in* Swetnam and Brown 2010). Scaling-up with a limited number of reference studies to the broader landscape can oversimplify fire history (Swetnam and Brown 2010). Any biases and assumptions related to the reconstruction methods may be amplified as well (Hessburg et al. 2007).

Previous underestimates of mixed-severity fire regimes occurred, in part, because studies applied techniques more suitable for reconstructing either low- or high-severity fire regimes (Hessburg et al. 2007; Amoroso et al. 2011; Heyerdahl et al. 2012). For example, techniques for estimation of fire frequency differ by fire severity (Amoroso et al. 2011). Low-severity regimes are dominated by surface fires that cause limited damage to canopy trees (Agee 1993). The resulting forests generally contain fire-scarred veteran trees making fire-scar analysis useful for estimating stand-level fire frequency (Agee 1993; Brown and Swetnam 1994). In high-severity regimes, infrequent, stand-replacing fires kill most canopy trees, initiating new even-aged stands (Agee 1993). In such forests, stands are mapped and years since the last fire are used to derive fire frequency metrics (Johnson and Gutsell 1994; Van Wagner et al. 2006). In contrast, mixed-severity regimes are characterized by a range of fire severities (low, moderate, high) and frequencies. Thus, a combination of research approaches is required as even-aged classes and fire scars are present at stand- and landscape-levels (Heyerdahl et al. 2012; Margolis and Balmat 2009).

Over the past 20 years, natural disturbance-based management has been implemented to achieve more sustainable harvesting practices that maintain biodiversity (Landres et al. 1999; Turner 2010). Moreover, destruction of communities and property by wildfires over the past decade has highlighted a critical need for landscape-level fire and fuel management plans (e.g., Stephens and Ruth 2005; Hirsch and Fuglem 2006). In British Columbia, these needs have been addressed using two classification systems developed to provide baselines of historic fire severity and frequency. The Natural Disturbance Type (NDT) system has been used to guide forest management for over two decades, but primarily emphasizes high-severity fire regimes (in 4 of 5 classes) (B.C. Ministry of Forests 1995). While frequent low-severity fire regimes are taken into account (in only one class), mixed-severity fire regimes are not. The Historic Natural Fire Regime (HNFR) system provides a more detailed classification system, which includes mixed-severity fire regimes (in 3 of 9 classes) and accounts for the influence of local topography on fire effects (Blackwell et al. 2003). Both systems assigned fire regime classes to landscape units (i.e., Biogeoclimatic zones, Table 2.1) using extrapolations from studies conducted at localized-scales or expert knowledge.

Given our poor understanding of mixed-severity regimes errors are likely present in existing fire regime classification systems. For example, in mid-elevation mountain forests of southeastern B.C., one classification system (NDT) describes high-severity fires averaging every 150 years; whereas the other system (HNFR) describes a mixed-severity regime with fire frequencies of < 35 to 100 years, depending on elevation, aspect and slope. Empirical tree-ring studies can be used to verify fire regime classification accuracy (e.g., Swetnam and Brown 2010); thus, we use dendrochronological methods to reconstruct the fire history of two watersheds across an elevation gradient encompassing a broad range of fire severity. We ask, *How well do existing fire-regime classification systems represent mixed-severity fire regimes?* We use fire scar and establishment dates of sampled trees to reconstruct fire history and classify past fire severity in 20 randomly-selected stands representative of the local elevation gradient. To evaluate the accuracy of the two fire regime classification systems, we compare their predictions to observed fire severity and two measures of fire frequency: fire return intervals (FRIs) and time since last fire (TSLF). Discrepancies between predictions and observations are addressed by critically examining underlying assumptions of each system. Lastly, we discuss the implications of our assessment for sustainable forestry and fire hazard mitigation in forests with mixed-severity fire regimes.

2.2 Study area

This research was conducted in the 15,400 ha Joseph and Gold Creek (JGC) watersheds (49°25’N 115°40’W) in the East Kootenays of southeastern British Columbia located directly south of the City of Cranbrook (Fig. 2.1). The watersheds are situated in the mid-elevations of the Rocky Mountain Trench on the leeside of the Purcell Mountain Range, within the greater Columbia River Basin. Climatic conditions are continental with a strong rainshadow effect created by the Purcell Mountains, which block the prevailing westerly air masses (Meidinger and Pojar 1991). Mean annual precipitation amounts to 383.4 mm with 71% in the form of rainfall at lower elevations (940 m.a.s.l., Environment Canada 2011). July average temperatures range from 10.9 ^{0}C to 25.6 ^{0}C (Environment Canada 2011). Parent materials and soils vary with elevation due to climate patterns and the many glacial advances and retreats during the last glaciation (Clague 1975; Valentine et al. 1978). Eutric Brunisol soils are most common at low- to mid-elevations where fluvioglacial gravel and morainal till deposits are found, while at higher-elevations, Humo-Ferric Podzols formed on coarse-grained materials including colluvium and till. Regosolic soils are discontinuous and occur where steep slopes and rocky outcrops result in shallow stony solums (Valentine et al. 1978).

The JGC watersheds extend from approximately 1000 to 2000 m.a.s.l. The dominant tree species include Douglas-fir (*Pseudotsuga menziesii*), western larch (*Larix occidentalis*), lodgepole pine (*Pinus contorta*), Engelmann spruce (*Picea engelmannii*) and subalpine fir (*Abies lasiocarpa*). The forests include the Dry Mild Interior Douglas-fir subzone (IDFdm), Dry Cool Montane Spruce subzone (MSdk), and Dry Cool Engelmann Spruce-Subalpine Fir subzone (ESSFdk) of the biogeoclimatic ecosystem classification (BEC) system of B.C. (Meidinger and Pojar 1991). Two Natural Disturbance Type (NDT) classes and three Historic Natural Fire Regime (HNFR) classes are present in different proportions of the JGC watersheds (Table 2.1). Since forests in the HNFR5 class were not accessible by road, they were not sampled (Da Silva 2009).

2.3 Materials and methods

Site selection

To represent the dominant forests of the Joseph and Gold Creek watersheds, a stratified-random approach was used to locate 20 sites, including 10 sites in each of the two dominant HNFR classes, HNFR2 and HNFR4 (Table 2.1). We used geographic information system (GIS) layers of Provincial forest cover to locate potential sites within stands that were dominated by conifers, unlogged, within 600 m of a road and greater than 3 ha (Da Silva 2009). Sites with evidence of early- to mid-20th century harvest (*c.*1950) were included if historical fires and stand dynamics could be reconstructed from intact stumps and remnant trees.

Fire scar records and stand age-structure

At each site, plots were placed at the center of the stand using GPS coordinates that were predetermined in the GIS. Elevation (meters above sea level, m.a.s.l.) was measured at plot center. Up to 10 fire-scarred trees were sampled in a 1-ha fire-history plot, although several sites did not have scarred trees. For plots with more than 10 fire-scarred trees, we selected the largest and presumably oldest trees, or trees with multiple scars to provide the most complete representation of plot-level fire history (Da Silva 2009). Fire scars were crossdated at an annual-level of resolution using standard dendrochronological methods (Brown and Swetnam 1994) to yield plot-level fire-scar chronologies (Da Silva 2009). In subsequent plot-level analyses, we used only fire-scar dates for years when ≥ 2 trees were present and could have been scarred.

Canopy and subcanopy trees were sampled at the center of each fire-history plot. To ensure old, canopy-dominant trees were not under-represented, we used an *N*-tree sampling scheme (Brown 2006) to select the 30 live trees, snags or stumps with dbh ≥ 25 cm that were closest to plot center. We tallied live subcanopy (dbh = 5 to 25 cm) trees by species in a 10 x 10 m plot. All canopy trees and a random subset of 10 subcanopy trees were sampled to assess tree ages. Increment cores were sampled as close to the base of trees as possible (30 ± 21 cm; mean ± standard deviation) to reduce age-estimate errors. Cores were visually assessed in the field to ensure they either intercepted or were close to the pith. Partial sections were removed from stumps sound enough to crossdate.

Increment cores and partial sections were prepared by mounting cores on wooden supports, gluing decayed partial sections, and sanding all samples using progressively finer paper (80 to 400 grit). Ring-width series of each sample were measured using a Velmex bench interfaced with a computer and statistically crossdated using the program COFECHA (Grissino-Mayer 2001). We used existing regional chronologies for the dominant species (L.D. Daniels unpublished data) and created new site- and species-specific chronologies to crossdate dead and partially decayed trees. For cores that did not intercept the pith (37%), we estimated the number of missing rings to pith using a geometric correction (Duncan 1989). The number of years for trees to grow to core height was estimated using height correction regressions representing different rates of growth (Wong and Lertzman 2001). Height-age regressions were developed by aging seedlings of different species at the root collar and up the stem at 10 cm intervals (L.D. Daniels unpublished data).

Site-level fire history interpretations

We used fire-scars to date low-severity fires and even-aged tree cohorts (hereafter 'cohorts') to date moderate- to high-severity fires which produce conditions conducive to tree establishment (Sherriff and Veblen 2006; Heyerdahl et al. 2012). Cohorts were identified using a sliding window to detect 20-year periods when a critical number of sampled trees established (pre-1800 ≥ 30 trees per ha, post-1800 ≥ 50 trees per ha) (Ehle and Baker 2003). We modified the criterion used to identify cohorts that established before 1800 since evidence of past fires depletes through time due to tree death, decay of coarse wood and subsequent fires (Ehle and Baker 2003). Cohorts that corresponded to a fire scar at the same site, or to a fire that scarred trees at ≥ 2 nearby sites, were assigned the same calendar year as the fire scars. Cohorts that did not coincide with a fire scar were assigned the calendar year prior to the pith date of the oldest tree in the cohort. We assumed all identified cohorts established as a result of fire, given three lines of evidence. (1) There were abundant fire scars in the study area, but individual sites lacked abundant logs or snags that could indicate stand-level windthrow or insect outbreaks. (2) Most cohorts included shade-intolerant lodgepole pine, which regenerate after fire in the study area. (3) Other recent research in southern British Columbia reported no evidence that tree cohorts were climatically forced (Heyerdahl et al. 2012).

The historical fire severity of each site through time was classified as 'mixed' or 'high' based on the presence or absence of fire scars and cohorts (Heyerdahl et al. 2012). A site with ≥ 2 different fire years indicated by scars and ≥ 1 cohort was classified as a mixed-severity fire history (Heyerdahl et al. 2012). Sites with cohorts present at the beginning of the age records and ≤ 1 fire scar were classified as a high-severity fire history (Antos and Parish

2002; Heyerdahl et al. 2012). The presence of fire scars and absence of cohorts would suggest a low-severity fire history (Heyerdahl et al. 2012); however, no sites in this study had these characteristics. We compared the mean elevations of sites classified as mixed- *versus* high-severity using a Student's *t*-test ($\alpha = 0.05$).

Fire return intervals (FRIs) and time since last fire (TSLF) were used to measure fire frequency using the composite fire records (all scars and cohort dates) for each site. FRIs, the number of years between consecutive fires at a site, were derived from the annually-resolved fire scar dates and estimated cohort dates. At sites where ≥ 3 fires had been dated, the mean and range of FRIs were calculated. Many sites did not have any fire scars; therefore, TSLF was calculated for all sites as the number of years between the most recent fire scar or cohort date and the year of sampling (i.e., 2009).

Comparative analysis: predicted vs. observed fire history

For each site, the HNFR and NDT classes were determined using GIS (Table 2.1). We compared fire severity and frequency predictions (based on HNFR and NDT classes) with observed values (based on field interpretations) for each individual site (Table 2.1). We grouped sites by NDT and HNFR classes (as listed in Table 2.1) to determine if observed fire severity corresponded with the prediction for each class. To evaluate fire frequency at the subset of sites with multiple fires, we compared mean observed FRIs from each site to predicted values. We determined the proportion of sites where observed fire intervals and observed mean FRI values were within the predicted ranges for a given class. Lastly, we compared the observed TSLF values to the predicted fire frequencies for each site and class (e.g., HNFR2/4, NDT4/3).

2.4 Results

Observed fire history

Fire histories of individual sites varied in duration and types of evidence of fire (Table 2.2, Fig. 2.2). A total of 155 fire scars were crossdated on 74 recorder trees located at 12 of 20 sites (Da Silva 2009, Table 2.2). The remaining eight sites lacked fire-scarred trees. We crossdated and determined establishment dates of 590 live trees and 64 dead trees (n = 33 ± 5 per site; mean ± standard deviation). At least one cohort was identified at each site (Table 2.2, Fig. 2.2). Across all sites, 37 unique fire years were identified between 1600 and 2009 based on direct evidence from fire scars (n = 31 years) and indirect evidence from even-aged tree cohorts only (n = 6 years) (Fig. 2.2). Fires in 1910, 1869 and 1721 were the most widespread in the study area (Fig. 2.2).

Eleven of 20 sites (55%) were classified as having mixed-severity fire histories; the remaining 9 sites (45%) were classified as high-severity fire histories (Table 2.2, Fig. 2.2). Mixed-severity sites included 2 to 15 fire years derived from fire-scarred trees and 1 to 4 even-aged cohorts (Table 2.2). For 8 of 12 cohorts, the inner-ring dates of the oldest tree corresponded with a fire-scar date at the site-level, suggesting that these trees established following a moderate-to-high severity fire (Table 2.2). The forests at sites classified as mixed-severity were primarily composed of fire-adapted tree species such as Douglas-fir, western larch and lodgepole pine, but a few sites also included Engelmann spruce and subalpine fir. At the high-severity sites, the lack of fire scars and presence of a single cohort including the oldest trees provided evidence of past stand-replacing fires. These cohorts were composed of lodgepole pine, with a Douglas-fir cohort at one site. Lodgepole pine, Engelmann spruce and subalpine fir dominated these sites, although Douglas-fir and western larch were sometimes present. On average, the mixed-severity sites were at significantly lower elevations (1374 ± 137 m.a.s.l.; mean ± standard deviation) than the high-severity sites (1752 ± 179 m.a.s.l.) ($p < 0.001$).

Fire frequency varied between mixed- and high-severity sites. Among the mixed-severity sites (n=11), intervals between fires ranged from 1–148 years, with the majority < 56 years and site-level mean FRIs were 7–139 years (Table 2.2). TSLF for the mixed-severity sites ranged from 56-104 years, with few recorded fires since the widespread 1910 fire, which burned at all mixed-severity sites (Fig. 2.2). TSLF was greater than historic maximum fire intervals at half of the mixed-severity sites and exceeded the mean and median FRI at all of these sites (Table 2.2). Because fire scars were absent at the high-severity sites (with the exception of site 17), FRIs could not be calculated. At these sites, TSLF ranged from 99 to 369 years, with the majority of sites exceeding 150 years since fire (5 of 9 sites; Table 2.2).

Fire severity: predicted vs. observed

At the site level, neither the HNFR nor NDT predictions corresponded well with observed fire severities and represented contrasting classification errors (Fig. 2.3). The HNFR predictions successfully identified 11 mixed-severity sites, but failed to identify any of the nine observed high-severity sites. In contrast, the NDT predictions correctly identified all nine high-severity sites, but misidentified nine mixed-severity sites as high-severity and two mixed-severity sites as low-severity.

At the class-level, the HNFR2 predictions represented fire severity more accurately than HNFR4 and both NDT classes (Fig. 2.3). The HNFR2 predictions accurately represented the mixed-severity regimes observed within this class (9 of 10 sites = mixed, 1 of 10 = high). In contrast, the same classification system failed to predict the high-severity regime prevalent in the HNFR4 class (8 of 10 sites = high; 2 of 10 = mixed). Conversely, for the NDT3 class the observed fire regime included a mix of severities (9 of 18 sites = high, 9 of 18 = mixed), yet the predicted regime was high-severity. In the NDT4 class, the observed regime was mixed-severity (2 of 2 sites = mixed), but the predicted regime was low-severity.

Fire return intervals: predicted vs. observed

The HNFR predictions were more successful than the NDT at classifying sites with frequent fires (i.e., mixed-severity sites with multiple fire scars) (Fig. 2.4). Observed site-level mean FRIs were generally within the predicted ranges of the HNFR classification and deviations were minor compared to errors in the NDT predictions. Sites in the HNFR2 class had mean FRIs of 7 to 56 years, which were similar to the predicted FRI range of 0 to 35 years (Fig. 2.4 A1). In the HNFR4 class, site-level mean FRIs ranged from 41 to 139 years across three sites, which generally corresponded with the predicted range of 35 to 100 years (Fig. 2.4 A2). In contrast, the 10 of 18 sites in the NDT3 class had mean FRIs of 7 to 56 years, which were less than the predicted mean FRI of 150 years (Fig. 2.4 B2). In fact, 85% of intervals were less than 50 years in length, suggesting these sites would have been more accurately represented by the NDT4 class with a predicted FRI range of 4 to 50 years. Lastly, the predictions for the NDT4 class were consistent with the observed mean FRIs, albeit for a limited number of sites (n=2; Fig. 2.4 B1).

High-severity sites with infrequent fires were more accurately represented by the NDT classification system than the HNFR (Fig. 2.4). Sites in the HNFR4 class had observed

TSLF values from 99 to 369 years, which were more consistent with the predicted mean fire frequency of 150 years for the NDT3 class than the predicted range of 35 to 100 years for the HNFR4 class. This difference is particularly pronounced as 50% of sites in the HNFR4 class have not burned in at least 150 years (Fig. 2.4 B2).

Neither classification system predicted the full range of variation in fire frequency observed at the site-level or among fire regime classes. In the HNFR2 class, only 67% of observed intervals were within the predicted range 0 to 35 years. The remaining intervals were between 35 to 100 years (26%) or greater than 100 years (6%) (Fig. 2.4 A1, n=53 intervals from 9 sites). Consequently, most sites (8 of 9) had one to four fire intervals that exceeded the predicted maximum FRI of 35 years (Fig. 2.4 A1). In the HNFR4 class, 4 of 5 observed fire intervals were within the predicted range (35 to 100 years). However, TSLF at all sites was ≥ 99 years, meaning the HNFR4 class did not predict the full range of variability found at these sites (Fig. 2.4 A2). In a similar way, the NDT3 class failed to predict the full range of intervals given that all observed intervals were lower than the predicted mean FRI of 150 years (Fig. 2.4 B2). In the NDT4 class, some intervals were > 50 years (11%), which exceeds the predicted FRI range of 4 to 50 years (Fig. 2.4 B1). However, high-severity fires are predicted to occur every 250 years (on average) in the NDT4 class and it is possible such fires occurred at shorter intervals at these sites (Fig. 2.4 B1 and B2).

2.5 Discussion

Mixed-severity fire regimes dominate at lower elevations

In mountainous terrain, mixed-severity fire regimes have been found to occur at lower elevations relative to high-severity regimes (Taylor and Skinner 1998; Schoennagel et al. 2004; Sherriff and Veblen 2006). Our results confirmed these patterns in the JGC watersheds where the historic fire regime varied similarly along a *c*. 800-m elevation gradient. Such patterns are in sharp contrast with existing fire regime classifications that guide forest management (B.C. Ministry of Forests 1995, Blackwell et al. 2003).

Low-to-moderate severity fires are found to be historically common in the mid-elevation forests of southeastern B.C. (Daniels et al. 2011). Over half of our study sites (55%) had two indicators of mixed-severity fire regimes: multiple fire-scarred trees and at least one post-fire cohort (Sherriff and Veblen 2006; Heyerdahl et al. 2012). At mixed-severity sites, fire frequency was highly variable. Site-level fire return intervals (FRIs) ranged from a few years to several decades and occasionally exceeding a century.

The remaining sites had characteristics of a high-severity fire regime, including a single post-fire cohort and, generally, no fire scars (Antos and Parish 2002; Heyerdahl et al. 2012). The absence of fire scars is not an appropriate sole indicator of high-severity fire (Falk et al. 2011); however, two additional lines of evidence support this claim. Firstly, the oldest trees at high-severity sites were cohorts of serotinous lodgepole pine (with the exception of one Douglas-fir cohort) (Antos and Parish 2002). Secondly, there were no other trees pre-dating these oldest cohorts. High serotiny (i.e., cones that require fire to germinate) in this region (Antos and Parish 2002) combined with the absence of veteran survivors supports our inference of high-severity fire. At high-severity sites, the current fire-free period (e.g., TSLF) ranged from 99 to 369 years, exceeding 250 years for over half the sites. In comparison, the current fire-free period at all mixed-severity sites was less than 99 years, although it is likely that 20th century fire exclusion may have lengthened this period at some sites (Da Silva 2009; Daniels et al. 2011).

Underestimating the prevalence of mixed-severity fire regimes in the NDT classification

The prevalence of mixed-severity fire regimes has been previously underestimated for many parts of North America (e.g., Sherriff and Veblen 2006; Hessburg et al. 2007) and B.C. is no exception. Despite its widespread use to guide forest management in B.C., the NDT classification system is simplistic in its representation of the fire history of mountain forests in two ways. First, it implicitly assumes historic fires were primarily high-severity stand-replacing events. Second, by ignoring frequent low-to-moderate severity fires, it overestimates the length of fire return intervals. The result is an underestimation of the proportion of the landscape where complex, mixed-severity fires occurred historically. In contrast to NDT predictions, our field evidence demonstrated high-severity fires were much less ubiquitous. Specifically, half of sites within the NDT3 class (i.e., high-severity fire regime) showed evidence of frequent mixed-severity fires.

Overestimating prevalence of mixed-severity regimes in high elevation locations

The HNFR classification system errs in the opposite direction, suggesting mixed-severity fire regimes were dominant throughout the entire JGC watersheds. However, we found infrequent stand-replacing fires dominating upper-elevations. Our results indicate a transition from a mixed- to a high-severity regime between c.1400–1650 m.a.s.l. The HNFR2 class (mixed-severity, high frequency fires) accurately identified a mixed-severity, high-frequency fire regime for 90% of sites at lower-elevation. This fire class did not account for the entire range of fire intervals, however. Over 30% of observed FRIs exceeded the maximum predicted return interval of 35 years. In contrast, within the HNFR4 class (predicting mixed-severity, moderate frequency fires), 80% of sites would have been more accurately described as having a high-severity and low-frequency fire regime based on observed field data.

Why do fire history classifications disagree on the same landscape?

A comparison of the assumptions underlying each classification system is helpful in explaining discrepancies between them as well as their lack of concordance with field results (Table 2.3). In addition, descriptive attributes of these classification systems (i.e., severity and frequency) are based on contrasting research methods with differing underlying assumptions (Table 2.3). Limitations of these contrasting research approaches have been the focus of several reviews (e.g., Johnson and Gutsell 1994; Lertzman et al. 1998; Veblen 2003). Thus, for our discussion, we emphasize the systematic mapping errors related to mixed-severity fire regimes. Methodological assumptions from empirical studies may translate into problematic landscape-level depictions of this fire regime type.

Traditional equilibrium models of succession are not a suitable theoretical framework for mixed-severity systems. For example, the NDT classification system implicitly assumes stand-replacing fires dominated fire regimes in all forests, except in valley bottoms (B.C. Ministry of Forests 1995). This erroneous assumption can be traced to the biogeoclimatic ecosystem classification (BEC), a fundamental component of the NDT system. The BEC system is based on the traditional view of succession in which stand-replacing disturbances initiate even-aged stands and succession proceeds in absence of disturbance until a climax forest develops (Meidinger and Pojar 1991). Given this underlying theoretical framework, it is understandable that stand-replacing disturbances are the primary focus of the NDT system (Table 2.3).

Analysis of ‘time-since-fire’ maps includes assumptions that are inconsistent with key attributes of mixed-severity fire regimes. Estimating fire frequency using time-since-fire maps requires delineating stands discernible with aerial photographs and assigning a single age to each stand (Johnson and Gutsell 1994; Van Wagner et al. 2006). This approach has been applied in the Canadian cordillera to estimate return intervals of 150 to 400 years for stand-replacing fires (Johnson et al. 1990; Masters 1990; Van Wagner et al. 2006). However, stand boundaries produced by low-to-moderate severity fires can be difficult to distinguish and the resulting stands are not even aged (Amoroso et al. 2011). In the NDT system, applying the results of ‘time-since-fire’ studies throughout the landscape scale has underestimated fire frequency and overestimated fire severity in forests that had mixed-severity fire regimes historically. Incorporating additional lines of evidence (e.g., Margolis et al. 2007) would help in differentiating mixed- and high-severity regimes.

In contrast to the NDT system, the HNFR classification system explicitly recognizes that historic fires burned at a range of severities and frequencies, creating mixed-severity fire regimes in some forests (Blackwell et al. 2003). Reference studies underpinning the HNFR system used fire scars from veteran trees to calculate fire return intervals for frequent low-to-moderate severity fires. These studies were conducted in mixed-conifer forests and estimated fire return intervals of *c.* 5 to 120 years at mid-elevations in southeastern British Columbia (e.g., Beck 1984; Gray et al. 2002), northeast Washington (e.g., Schellhaas et al. 2000), and western Montana and Idaho (e.g., Arno 1976; Davis 1980; Barrett et al. 1991). These studies generally report higher fire frequencies with decreasing elevation and on warmer aspects, two spatial patterns reflected in the HNFR system.

When extrapolating metrics from stand-level fire-scar studies, both site selection and research design must be considered, especially to avoid overestimating the occurrence of low-to-moderate severity fires at the landscape-level. It appears two such factors were at issue in HNFR classification system. First, fire-scar evidence is not uniform across the landscape. Thus, studies using only sites with fire-scarred trees represent only a subset of the landscape where low-to-moderate severity fires burned (Veblen 2003). Direct extrapolation of fire history statistics from fire-scar sites to the broader landscape, without consideration of sites lacking fire scars, can overestimate fire frequency. Second, frequency estimates depend on the size of the sampled area (Johnson and Gutsell 1994). In general, with increasing plot size or when multiple plots are combined, fire frequency estimates increase and fire return intervals decrease. For example, fire statistics based on all sites in a landscape generally indicate higher frequency and shorter fire return intervals relative to individual sites within the same record. In the JGC watersheds, the HNFR classification may have overemphasized low-to-moderate severity fires because it was based on studies conducted only at sites with fire-scarred trees (e.g., Arno 1976, Davis 1980, Beck 1984). Furthermore, statistics from fire records that combined data from two to nine plots were extrapolated to the landscape-scale (e.g., Gray et al. 2002) (Table 2.3).

Diversifying forest management to emulate mixed-severity regimes

Natural-disturbance-based management has been adopted by many jurisdictions as an ecologically-sustainable approach to managing forests (e.g., B.C. Ministry of Forests 1995; Cissel et al. 1999). When based on the assumption that high-severity fires are the dominant disturbance type, emulation of fire regimes is often accomplished using even-aged silvicultural systems (Table 2.3). For instance, conventional clear-cut harvesting (with rotations of 80 to 100 years) is used to emulate infrequent stand-replacing disturbances (Bergeron et al. 2002). However, it is questionable whether conventional clearcut harvesting suitably emulates stand-replacing fires, which are more heterogeneous than traditionally thought (Bergeron et al. 2002; Turner 2010). Moreover, clearcut harvesting does not emulate mixed-severity fire regimes, in several important aspects.

Emulation of mixed-severity fire regimes would require a variety of silvicultural approaches across the landscape, particularly those that maintain biological legacies and forest structural heterogeneity (Franklin et al. 2002). In mixed-severity regimes, recurring low-to-moderate severity fires create and maintain open multi-aged stands with a diversity of structural features important to ecosystem function and landscape connectivity (Perry et al. 2011). Large veteran trees and snags provide essential habitat for wildlife (e.g., Spies et al. 2006). Silvicultural systems that include heterogeneity within and among cutblocks would be more consistent with the effects of low-to-moderate severity fires. Green-tree retention of large dominant trees could represent features of unburned patches or individual fire-tolerant trees. These could potentially offset losses of large diameter trees due to high-grading seen in many regions throughout the early 20th century (e.g., Hessburg et al. 2000).

Estimating fire hazard: a public safety priority

In recent decades, wildfires have burned an average of 2.5 million ha year^{-1} in Canada and efforts at fire suppression approach $1 billion annually in the US and Canada (Stephens and Ruth 2005; Hirsch and Fuglem 2006; Donovan and Brown 2007). The HNFR approach was intended to assess catastrophic fire risk in B.C. due to fuel accumulation resulting from fire exclusion (Blackwell et al. 2003), a problem in many western forests (Stephens and Ruth

2005; Donovan and Brown 2007). Limited resources require focusing efforts where fire exclusion has had the greatest impact. Generally, impacts are most pronounced in forests characterized by frequent low-to-moderate severity fires which are the easiest to suppress using modern fire-fighting techniques (Schoennagel et al. 2004). In mixed-severity regimes, however, the impact of fire suppression may be quite heterogeneous. Fire occurrence has decreased in our study area due to land-use practices and active fire suppression over the past 70 years (Daniels et al. 2011). Many of our mixed-severity sites had not recorded a fire since the widespread 1910 burn, yet the current fire-free interval exceeded the historic maximum FRI at only half of these sites. It is less likely fire exclusion significant impacted high-elevation forests and any old stands (TSLF > 250 years) are still within the historic range of variability. Thus, by overestimating the prevalence of mixed-severity regimes for JGC, the HNFR predictions may also inflate the spatial extent and severity of the fuel build-up problem (Table 2.3).

With an emphasis on mid-elevation mountain forests, we answered the important question of, "*How well do existing fire-regime classification systems represent mixed-severity fire regimes?*" We determined the answer is, in fact, not very well by several specific measures. Compared to the NDT system used to guide natural-disturbance-based management, only 10% of fire severity predictions agreed with our observations at lower-elevations (<1500 m.a.s.l.). For the HNFR system used to identify forests with increased fuel hazard, over-representation of mixed-severity regimes at higher-elevations likely over-estimates the extent and severity of fuel build-up. Occupying approximately 10% of Canada's landbase, as well as a large portion of the North American Cordillera, management of mixed-severity regimes in B.C. has far-reaching implications for a large proportion of mountain forests of western North America. As such, it is imperative that maps, classification systems, and their underlying assumptions represent the best available science and are updated to include emerging knowledge regarding the prevalence of mixed-severity regimes.

Table 2.1 Summary of fire-regime classification systems and class attributes represented in the Joseph and Gold Creek watersheds.

Classification system	Class	Fire severity regime	Fire return interval (range or mean)	Biogeoclimatic (BEC) subzone	Proportion of study area	Number of study sites (n)	GIS data source
Historic Natural Fire Regime (HNFR)	HNFR2	Mixed	0 to 35 yrs	IDFdm MSdk	30%	10	B.C. Forest Analysis and Inventory 2012
	HNFR4	Mixed	35 to 100 yrs	MSdk ESSFdk	57%	10	
	HNFR5	High	35 to 100 yrs	ESSFdk	12%	0	
Natural Disturbance Type (NDT)	NDT4	Low*	4 to 50 yrs*	IDFdm	4%	2	Blackwell et al. 2003
	NDT3	High	150 yrs	MSdk ESSFdk	96%	18	

*This NDT class also predicts that high severity (stand replacing) fires occurred every 150 to 250 years (or greater).

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Table 2.2 Comparison of site-level fire histories derived from fire scars and even-aged tree cohorts with associated fire regime classes from the Historical Natural Fire Regime (HNFR) and Natural Disturbance Type (NDT) classification systems. Sites are arranged from low to high elevation.

Site Attributes		Observed Fire History						Predicted Fire History	
#	Elevation (m.a.s.l.)	Number of scarred trees	Number of fire years	Number of cohorts	Fire severity	Fire return interval (years) mean (range)	Time since last fire (years)	HNFR classes	NDT classes
1	1168	9	6	1	Mixed	37 (12 – 92)	87	2	4
2	1259	7	6	1	Mixed	14 (10 – 133)	99	2	3
3	1287	7	15	1	Mixed	24 (4 – 91)	68	2	4
4	1320	8	5	1	Mixed	38 (11 – 91)	99	2	3
5	1330	6	6	4	Mixed	47 (4 – 148)	99	2	3
6	1352	8	4	1	Mixed	28 (21 – 41)	99	2	3
7	1366	9	9	2	Mixed	32 (4 – 50)	99	2	3
8	1386	3	3	1	Mixed	7 (1 – 13)	85	2	3
9	1417	0	0	1	High	--	99	2	3
10	1420	7	7	2	Mixed	44 (5 – 108)	56	2	3
11	1570	0	0	1	High	--	288	4	3
12	1581	3	2	2	Mixed	(41)	99	4	3
13	1645	6	3	2	Mixed	56 (41 – 71)	99	4	3
14	1668	0	0	1	High	--	140	4	3
15	1739	0	0	1	High	--	288	4	3
16	1746	0	0	1	High	--	99	4	3
17	1871	1	1	1	High	(139)	104	4	3
18	1872	0	0	1	High	--	350	4	3
19	1905	0	0	1	High	--	369	4	3
20	1981	0	0	1	High	--	274	4	3

Table 2.3 Comparison of the assumptions which explain the discrepancies between two fire regime classification systems and their implications for management.

Attributes compared	Classification system	
	Natural Disturbance Types (NDT)	**Historic Natural Fire Regime (HNFR)**
Underlying framework		
Theoretical foundation	Traditional, equilibrium successional paradigm	Contemporary, non-equilibrium successional paradigm
Fire regimes represented at mid- and high-elevations	High severity Frequency = 150 to 350 years	Low, mixed and high severity, Frequency = 0 to >200 years
Research approach and metrics	Time-since-fire (or stand-origin) maps with age-class analysis	Fire-scar interval analysis
Temporal fire metrics	Stand-level fire frequency and landscape-level fire cycle	Stand-level fire frequency and return interval
Limitations of research approach	Assumes stand-replacing fires, poor detection of lower severity fires	Estimates sensitive to research design and scale of analysis
Discrepancies between predicted and observed fire regimes		
Comparison with empirical data from JCG watersheds	Overestimates fire severity and underestimates fire frequency at mid-elevations	Underestimates fire severity and overestimates fire frequency at high-elevations
Forest management implications		
Disturbance-based approach	Traditional, even-aged silvicultural systems	Alternative silvicultural systems that retain structure and biological legacies
Effects on forest diversity	Homogenizes stand and landscape age structures	Adds heterogeneity to stand ages and size structures, diversifies landscape
Conservation of biodiversity	Reduction and loss of structurally diverse stands and habitats	Maintenance and restoration of structurally complex stands and habitats
Wildfire management implications		
Fire hazard assessment	Underestimates fuel and fire hazards at mid-elevations	Over-estimates fuel and fire hazards at high-elevations
Restoration and mitigation needs	No recognized need at mid-elevations	Identified need at mid-elevations and, potentially at high-elevations

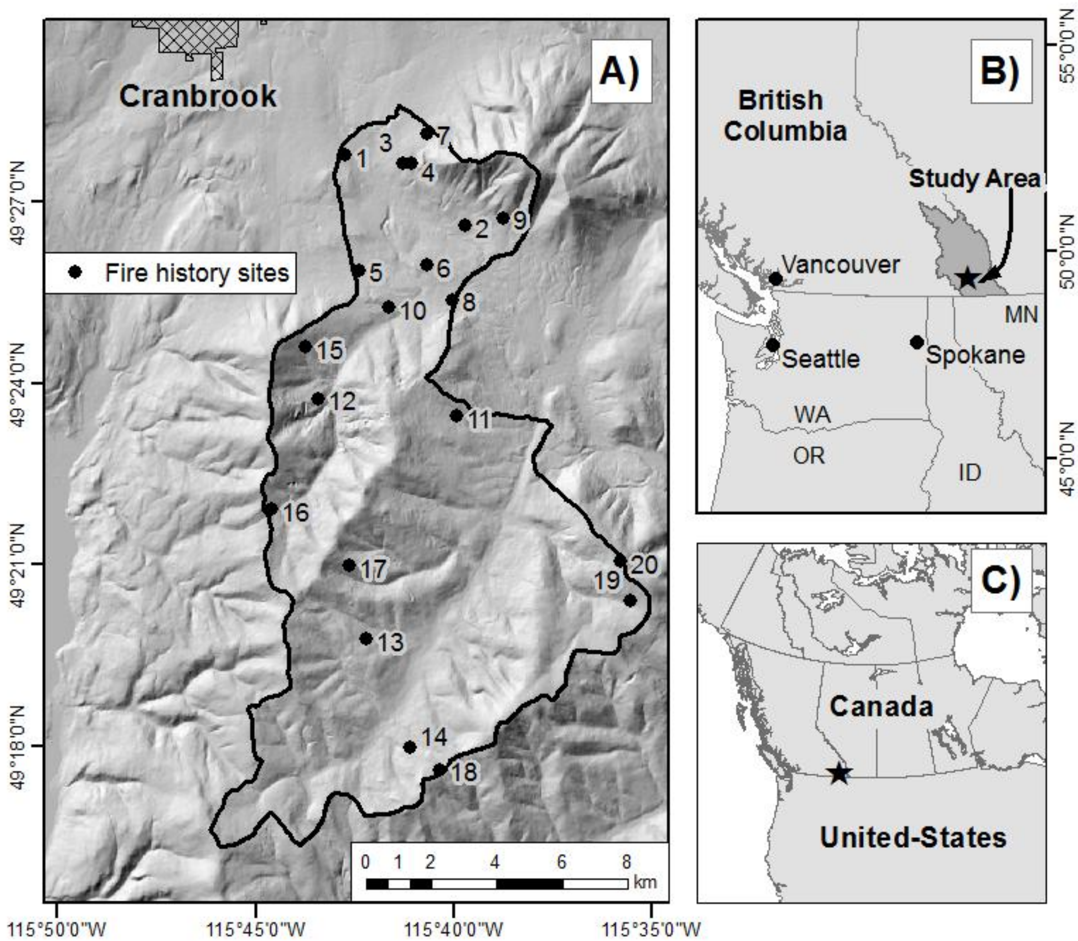


Fig. 2.1 (A) Location of the 20 research sites in the Joseph and Gold Creek watersheds, which provide the drinking water supply for the City of Cranbrook. (B) Location of the watersheds (star) within the Rocky Mountain Forest District (dark grey) of southeastern British Columbia, Canada. (C) Location of the watersheds within the context of North America.

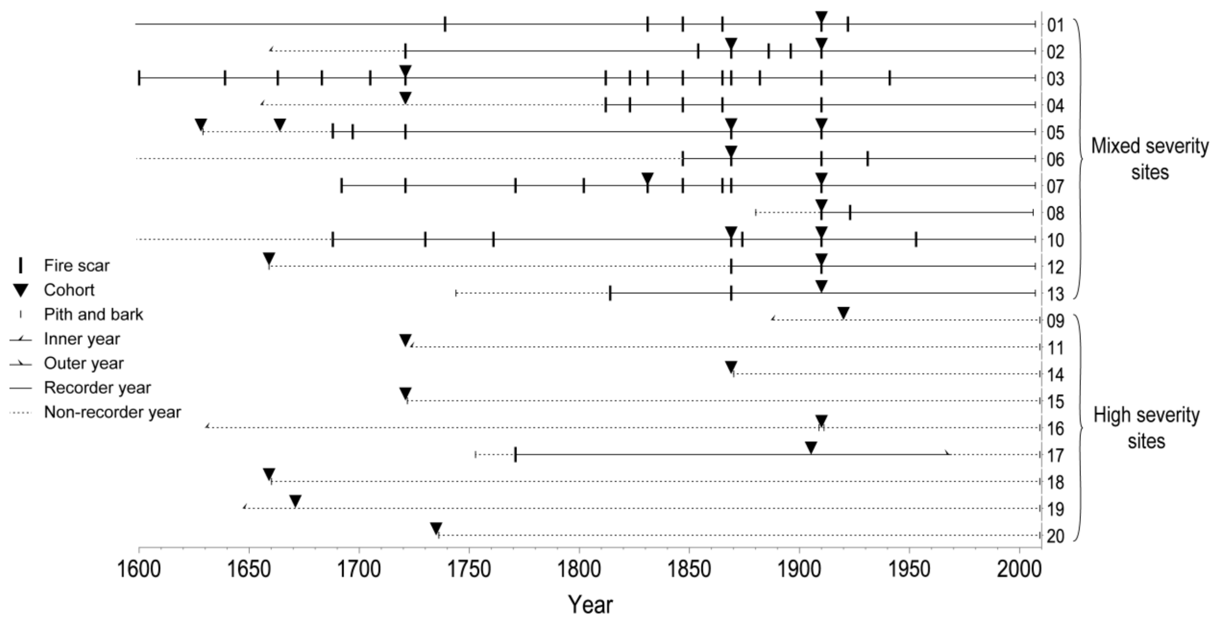


Fig. 2.2 Fire occurrence from 1600 to 2009 at sites classified as having mixed-severity (n = 11, top) or high-severity (n = 9, bottom) fire histories in the Joseph and Gold Creek watersheds. Horizontal lines represent the composite tree-ring record at each site. The length of each line represents the period of record, starting at the pith or inner ring of the oldest tree to the bark or outer ring of the sampled trees. Fire evidence includes crossdated, annually-accurate fire-scar dates and the initial date assigned to even-aged cohorts. For sites with fire scars, recorder years are those following the formation of the first scar through the year of sampling or year of death of sampled trees. For sites lacking fire scars, non-recorder years indicate the length of the tree-ring record.

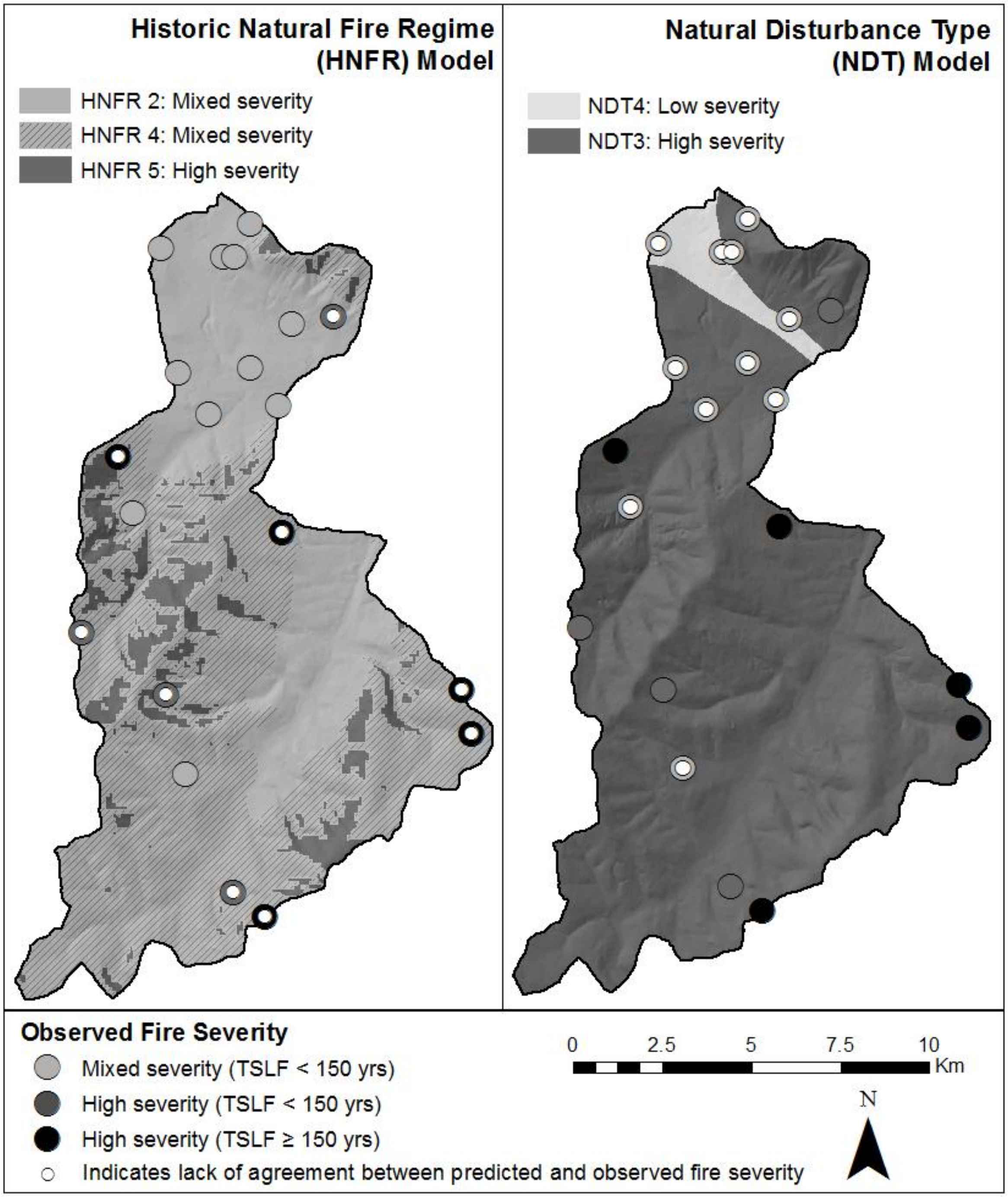


Fig. 2.3 Comparison of predicted versus observed site-level fire severities. Background shading shows predicted fire severity classes for the HNFR (left) and NDT (right) models. Large circles denote observed fire severity determined from fire scars and age cohorts; high-severity sites are differentiated according to time since last fire. Small white circles inset in the large circles denote sites for which predictions of fire severity were incorrect.

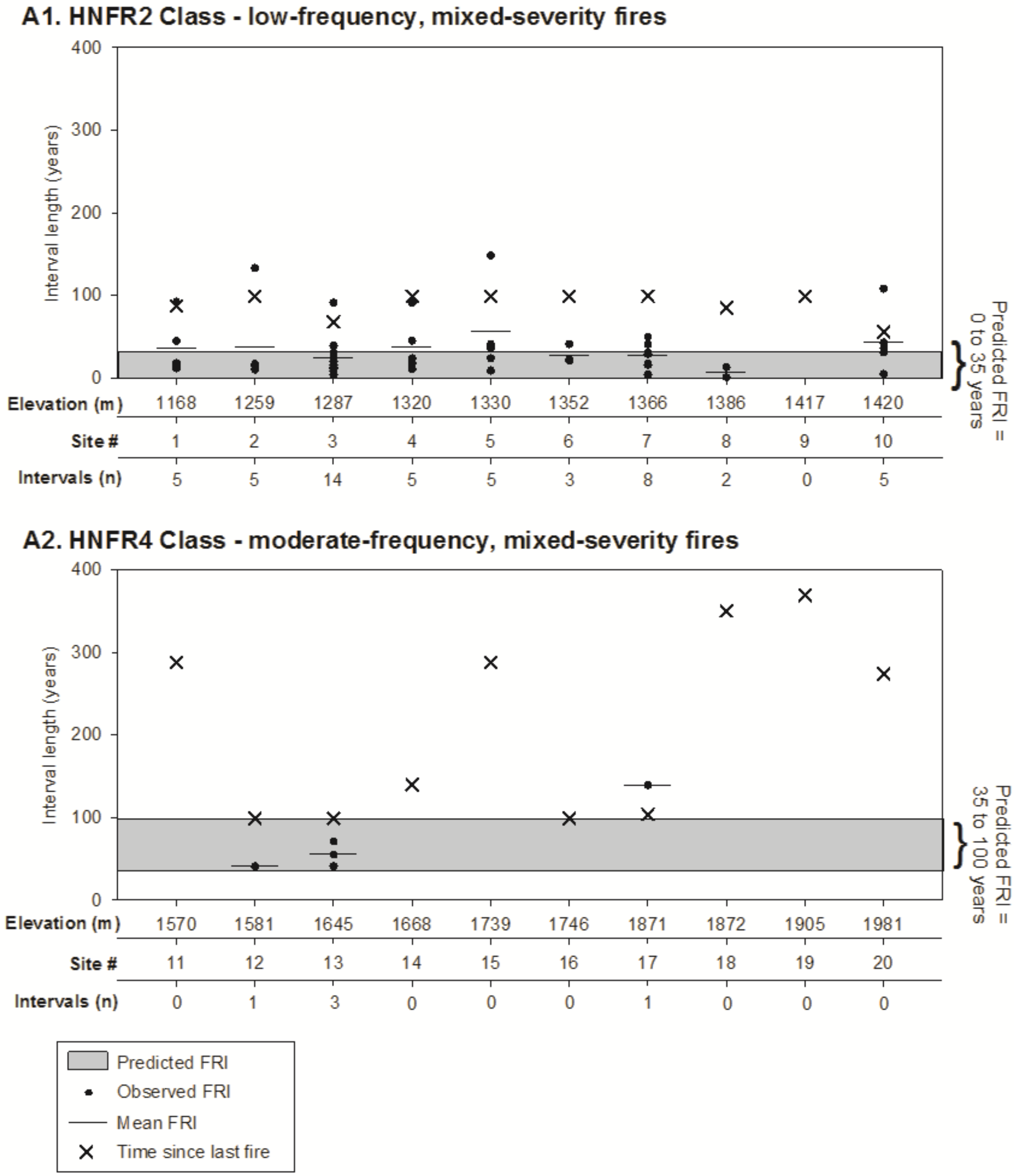


Fig. 2.4 Comparison of predicted and observed site-level fire frequencies. Sites are grouped according to HNFR (Panels A1/A2) and NDT (Panels B1/B2) classes. Within each panel, sites are arranged from low (left) to high (right) elevation. For each class, grey shaded areas represent predicted fire return intervals (FRIs, range or mean). Observed FRIs and time since last fire (TSLF) were reconstructed at each site using fire scars and even-aged cohorts.

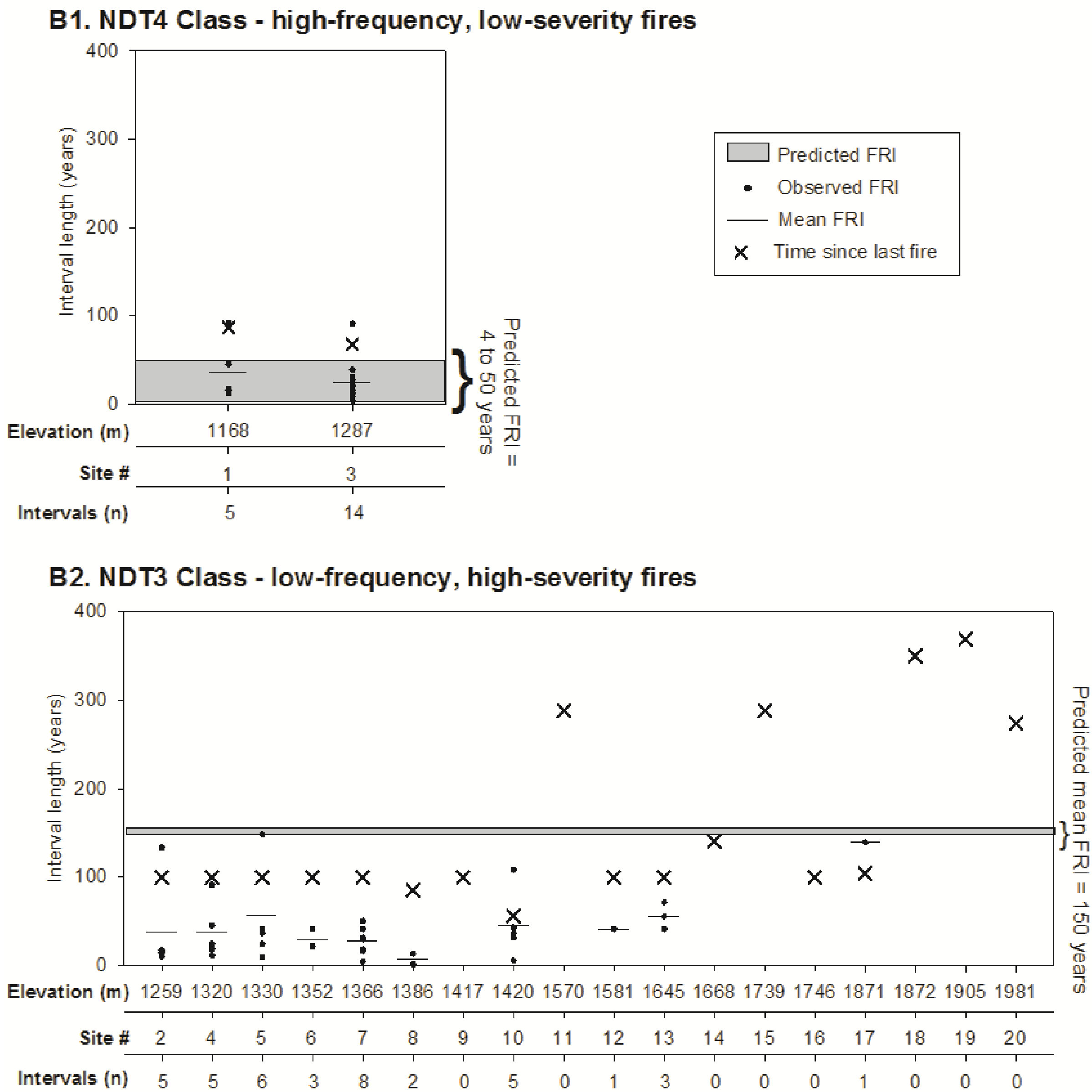

B1. NDT4 Class - high-frequency, low-severity fires
Interval length (years)
400
300
200
100
0
Elevation (m) 1168 1287
Site # 1 3
Intervals (n) 5 14
Predicted FRI = 4 to 50 years
Predicted FRI
Observed FRI
Mean FRI
Time since last fire
B2. NDT3 Class - low-frequency, high-severity fires
Interval length (years)
400
300
200
100
0
Elevation (m) 1259 1320 1330 1352 1366 1386 1417 1420 1570 1581 1645 1668 1739 1746 1871 1872 1905 1981
Site # 2 4 5 6 7 8 9 10 11 12 13 14 15 16 17 18 19 20
Intervals (n) 5 5 6 3 8 2 0 5 0 1 3 0 0 0 1 0 0 0
Predicted mean FRI = 150 years

Fig. 2.4 Con't.

Chapter 3: Stand dynamics and forest structure in mixed-severity fire regimes of interior mountain forests

3.1 Introduction

Increasing recognition of the historical prevalence of mixed-severity fire regimes has resulted in greater appreciation of their role in creating and maintaining heterogeneity in forested landscapes (Hessburg et al. 2007; Brown et al. 2008; Perry et al. 2010; Halofsky et al. 2011). Until recently, most research focused on quantifying fire *frequency* with less attention paid to fire *severity*. Consequently, severity was often assumed based on forest composition (e.g., low-severity in ponderosa pine (*Pinus ponderosae*) forests, high-severity in lodgepole pine (*Pinus contorta*) or mixed-conifer forests). The wide variability in severity within forest types and among different regions has prompted increasing emphasis on direct estimation of severity, as well as re-evaluation of fire histories in forests typified as low- or high-severity systems. This re-evaluation in Douglas-fir (*Pseudotsuga menziesii*) - ponderosa pine forests (often considered an exemplar of low-severity regimes) has yielded surprising evidence of mixed-severity fire histories in several regions including the Colorado Front Range (e.g., Sherriff and Veblen 2006) and southern interior of British Columbia (e.g., Heyerdahl et al. 2012).

Fire regimes are generated by interactions between regional climate (e.g., precipitation, temperature) and local factors that influence the flammability of fuels (e.g., topography, vegetation). In mixed-severity regimes, of particular importance are local environmental gradients which can generate heterogeneity within and among fires. In many regions, fire severity patterns vary along elevation gradients where mixed-severity regimes occur between low- and high-severity regimes (Schoennagel et al. 2004; Sherriff and Veblen 2006; Margolis and Balmat 2009). Aspect may also influence fire severity, with more severe fires on cooler northerly aspects relative to adjacent southerly aspects (Taylor and Skinner 1998). Forest structure influences wildfire behavior by affecting flammability (e.g., dense stands, ladder fuels, low canopy heights) (Schoennagel et al. 2004). Some structural attributes are species-specific (e.g., shade tolerant species have more ladder fuels and lower canopy heights relative to shade intolerant species) (Perry et al., 2011). In turn, species composition and structure may also be influenced by fire severity (Agee 1993). Other disturbance agents, and autogenic processes (i.e., succession, self-thinning) may also play a role in fire regime dynamics (Antos and Parish 2002; Wimberley and Spies 2001; Sinton et al. 2000), although their contribution to fire behavior are poorly understood.

Improving management of forests with mixed-severity regimes is a research and management imperative (Spies et al. 2006; Perry et al. 2011). Knowledge of forest attributes at the landscape- (e.g., patch sizes) and stand-level (e.g., veteran trees, size classes, snag density) is a key requirement for silvicultural practices that aim to emulate natural patterns (Keane et al. 2009). Mixed-severity regimes are thought to produce landscape patterns distinct from low- and high-severity systems (Perry et al. 2011) including a greater number of small-sized patches and few large patches, resulting in higher edge to interior ratios (Hessburg et al. 2007; Collins and Stephens 2010). The proportion of low- vs. high-severity patches also characterizes mixed-severity regimes among regions (Brown et al. 2008; Perry et al. 2011).

In contrast, our knowledge of stand-level attributes in mixed-severity systems is fairly limited and as a result influenced by untested assumptions and hypotheses. One assumption is that complex age-structures of mixed-severity systems lead to more complex stand structures (e.g., Taylor and Skinner 2003); whereas simpler age- and stand-structures are produced by fires of high-severity. It has been hypothesized that fire-prone forests (e.g., low- and mixed-severity) may harbor stand structural attributes similar to classical old-growth forests that develop in the absence of disturbance (Spies et al. 2006; Perry et al. 2011). Unfortunately, comparisons of stand-level attributes generated by mixed- and high-severity fire regimes within the same landscape have rarely been attempted to date. Because such a comparison would yield important insights for forest management, this research addresses this critical knowledge gap.

Research objectives

My primary goals are to improve our knowledge of how stand dynamics and forest structure differ between forests affected by high- versus mixed-severity fires and to understand the role of topography in influencing patterns of fire severity. Using a fire history dataset from interior mountain forests of British Columbia, I address two questions: (1) *What is the prevalence of mixed-severity fire across the landscape and how is it influenced by topography?* (2) *How do fire and topography interact to influence forest composition and structure?* To answer these questions, I used 20 randomly-selected sites and fire history data (from Chapter 2), with the addition of new site-level data which includes: forest age-structure information, size classes, snag and veteran tree densities, as well as topographic characteristics.

A secondary goal of this work is to compare two different approaches used to infer severity of past fires as approaches vary among different published studies.

Dendrochronological methods are often used to infer severity for past events where direct measurements (e.g., pre- and post-fire, Thompson and Spies 2010) are impossible to obtain. One approach, applied in Chapter 2, uses fire scars as evidence of low-severity fires and the presence of even-aged cohorts to indicate high-severity fire (Brown et al. 1999; Heyerdahl et al. 2012). Presence of both fire scars and even-aged cohorts has been used as an indicator of a mixed-severity regime at the site-level (Chapter 2, Heyerdahl et al. 2012). A second approach is based on the number of different age-classes, with the assumption that stands with high-severity fire have fewer age-classes than those characterized by less severe fires (Taylor and Skinner 2003). Few studies have evaluated the concurrence of these two approaches. As a result, it is important that the scientific community's re-examination of mixed-severity regimes also includes a rigorous evaluation of the techniques used in determining fire severity.

3.2 Study area

Field research was conducted in the 15,400 ha Joseph and Gold Creek (JGC) watersheds (49°25'N 115°40'W), situated directly southeast of the City of Cranbrook, located within the greater Columbia River Basin of southeastern British Columbia. The watersheds span approximately 1000 to 2000 m.a.s.l. on the leeside of the Purcell Mountain Range and upslope from the Rocky Mountain Trench.

The study area has a diverse land use history. It lies within the Ktunaxa Nation traditional territory. Although fire was used by the Ktunaxa people to enhance the growth of important plant species (e.g., bitterroot (*Lewisia rediviva*) (Mah 2000), to improve hunting and maintain open stands for campsites and travel (Barrett 1981, Barrett and Arno 1982), its use was likely limited to lower elevations. European settlement began in the late 1800s. Timber harvesting, cattle grazing and forest management have been predominant in the watershed since the early 20th century. Today, the Joseph and Gold Creek Watersheds provide the drinking water source for Cranbrook, B.C., and makes up part of the wildland-urban interface for this community.

3.3 Methods

Fire and disturbance history

Study sites (n=20) were selected using a stratified-random design with 10 sites located within one of two pre-existing fire regime classes (HNFR2 and HNFR4, Chapter 2) (Fig. 3.1). Three plot types were nested at the center of each site: fire scar (Da Silva 2009, and as described in Chapter 2), canopy-dominant tree (live or dead, dbh ≥ 25 cm), and subcanopy tree (live only, dbh = 5−25 cm) (Chapter 2) (Table 3.1). Species and dbh were recorded for each tree. All canopy trees and a random subset of 10 subcanopy trees were cored for establishment dates (Table 3.1). Partial sections were taken from dead trees including snags and stumps sound enough to sample. I identified any evidence of mountain pine beetle mortality in *Pinus contorta* snags based on visible entrance holes in the bark, pitch tubes, blue stain fungus in the sapwood and *J*-shaped galleries in the cambial tissue (Axelson et al. 2009). Death dates of snags and harvest dates of stumps were determined when samples had intact outer-rings and bark. Veteran trees were identified at each site as canopy trees (dbh ≥ 25 cm) that pre-dated and survived at least one fire (Table 3.1). The methods for sample preparation, fire scar dating and age estimates are described in Chapter 2.

I assigned each site to one of three fire history groups (I. Mixed severity, young forest; II. High severity, young forest; and III. High severity, old forest). Two site-level criteria (Table 3.2, modified from Chapter 2) were used to differentiate fire history groups: (1) the presence or absence of fire scars on living trees; and (2) time since last fire (TSLF). The presence or absence of fire scars was used as evidence of low-to-moderate severity fires. The presence of fire scars (along with 'cohorts', defined in Chapter 2) is indicative of a mixed-severity fire regime, while the absence of fire scars (but presence of a 'cohort') would suggest a high-severity fire regime. Time since last fire (TSLF) is indicative of forests undergoing different stand development processes following stand-replacing disturbances. Thus, for high-severity regimes I differentiated stands in which TSLF was <150 *versus* ≥150 years. The 150-year threshold is based on the mean fire-return interval estimated for the study area (B.C. Ministry of Forests 1995). Methods used to determine TSLF are explained in Chapter 2.

Topography

Topographic characteristics were obtained in the field at the center of each site (Table 3.1). Elevation was determined using a GPS. Slope aspect and steepness were measured using a compass and clinometer. Slope aspect was converted into a linear index representing warm (0 = southwest) to cool (180 = northeast) aspects.

Forest composition and structure

Species-specific density and basal area (per hectare) were calculated for each plot. For the canopy tree plot, a Poisson distribution adjustment was applied to correct for biases since trees were sampled using a plotless distance method (Lynch and Rusydi 1999). To calculate the basal area of stumps, I reconstructed stump diameter at breast height using regression models for trees in British Columbia (Demaerschalk and Omule 1982). I calculated species-specific importance values for each site as the sum of relative basal area and relative density (index value between 0-200) of all live trees and stumps with dbh ≥ 5 cm (Table 3.1).

To assess the influence fire has on tree establishment, I determined the lag of establishment for each individual tree following low-to-moderate or high-severity fire (Table 3.1). At each site, lags were calculated for each tree as the difference between the date of the last fire and subsequent tree establishment. I differentiated establishment lags between high-severity fires, indicated by a cohort versus low-to-moderate severity fires, indicated by fire scars. Lags were grouped by species and frequency histograms were visually assessed.

To address differences in structural attributes among the fire history groups, I grouped all living trees by three dbh size classes (5-24.9 cm, 25-44.9 cm, ≥45 cm) (Table 3.1). I calculated the densities of snags, excluding snags of lodgepole pine that were recently killed by mountain pine beetle (Table 3.1). To represent tree size variability, the coefficient of variation of dbh for trees ≥25 cm dbh and maximum dbh were calculated for each site.

Age-structure

I derived two age-structure indices to differentiate the effects of fire severity on stand dynamics among fire history groups (Table 3.1). The first index, evenness, represented whether each site was relatively even- or uneven-aged by counting the number of 15-year age classes occupied by at least one tree. Even-aged sites have fewer classes (narrower range of ages) than uneven-aged sites. The second index, continuity, represented whether age-structures were relatively continuous or discontinuous by counting the number of unoccupied (or empty) 15-yr age classes within the range of occupied classes. Continuous age structures have fewer unoccupied classes than discontinuous sites.

Statistical analysis

To address Question 1 on topographic effects on firet, I used fire demography diagrams for each site to evaluate the correspondence of tree establishment with fire scar and cohort dates. I also tested for differences in topographical site characteristics among the fire history groups using ANOVA and a post-hoc Tukey test (Kutner et al. 2005).

To address Question 2 on fire and topographic effects on forest composition and structure, I used importance valuesp to test for species compositional differences among fire history groups using a Kruskal-Wallis H-Test and pairwise comparisons with Bonferroni correction (Kutner et al. 2005) in R software and asbio package (Aho 2012). I also tested for differences in forest structural attributes among the fire history groups using this same approach. I used principal components analysis (PCA) to relate variation in species composition to topography and stand structure (Beaty and Taylor 2007). PCA was used to ordinate the species importance values from each site in R software using the vegan package (Oksanen et al. 2011). Compositional gradients were then identified by correlating the original species importance values (i.e., component loadings) with the three PCA axis scores (Peres-Neto et al. 2003). Topographic characteristics and structural attributes were correlated with PCA axis scores to identify underlying trends. Correlation significance was determined using a cut-off value of |0.5| (Richman 1988).

To examine the concordance among approaches used to infer severity of past fires, I compared the two age-structure indices among the three fire history groups using ANOVA and a post-hoc Tukey test (Kutner et al. 2005). It was expected that study sites affected by mixed-severity fires (Group I) would be more uneven-aged and discontinuous than sites burned by recent high-severity fires (Group II). The sites that burned at high severity but with a long TSLF (Group III) would be variable due to post-fire stand development processes.

3.4 Results

Fire history

Tree ages and establishment dates were determined for 590 live trees and 64 dead trees, which included 45 snags and 19 stumps. On average, 33 ± 5 (mean ± standard deviation) tree ages were determined for each of the 20 sites. Of the cores sampled, 63% intercepted with the pith. The combined error from both the pith and height corrections was 9 ± 9 years; 87% of corrections were less than 15 years.

Each of the 20 sites had evidence of past fire events. Fire scars were present on live trees at 11 sites and provided evidence for 6 ± 4 fire years per site (Table 3.3, Fig. 3.2a). These 11 sites also had evidence of 2 ± 1 cohorts per site, with 74% of the cohort dates corresponding with fire years denoted by a scar. The remaining nine sites had a single even-aged cohort including the oldest trees in the stand (Table 3.3, Fig. 3.2b and 3.2c). Fires years recorded at ≥ 4 sites included the 1721, 1847, 1865, 1869 and 1910 fires. Among all recorded fires, the 1910 fire was detected at the greatest number of sites (12 of 20).

TSLF ranged from 56 to 369 years (Table 3.3, Fig. 3.2). While 75% of the sites burned in the last 150 years, a quarter of the sites have not burned in over 250 years. Only four fires were recorded among the sites since the widespread 1910 fire. Unlike many fires of the 1800s that were recorded at multiple sites, the four fires after 1910 were detected at single sites.

Comparisons among fire history groups

Most sites (n = 11) were assigned to Group I (Mixed-severity fire, young forest), while the remaining nine sites were assigned to either high-severity fire groups (Table 3.3). All Group I sites burned during the 1910 fire, but had multiple fire scars that pre-dated 1910 and veteran trees that accounted for a large proportion of trees (dbh ≥ 25 cm). TSLF ranged between 56 and 99 years, but the oldest veterans originated in the 1300s, or during the 1600 to 1700s. Four sites were assigned to Group II (High-severity fire, young forest). Some of sites last burned during fires in 1910 and 1865 like some of the sites in Group I, but other fire years (1905, 1920) were unique to this group. TSLF ranged from 85 to 144 years and veteran trees were absent except at Site 9. The other five sites were assigned to Group III (High-severity fire, old forest). These sites last burned 274 to 350 years ago (Table 3.3) and no veteran trees were present.

Age-structure indices varied significantly among the three fire history groups (Table 3.4). Groups I and III were more uneven-aged than Group II. Specifically, Groups I and III had greater mean number of occupied 15-year age classes than Group II (p = 0.046 and p = 0.015, respectively). Group I had the most discontinuous age structures. It had the greatest mean number of unoccupied 15-yr age classes relative to Groups II and III (p = 0.005 and p =

0.04, respectively). Group II had the least unoccupied classes and most continuous age structures, but were not significantly different from Group III ($p = 0.58$).

Mean elevation was significantly different among the three fire history groups ($p = <0.001$) (Fig. 3.3). Group I had a significantly lower mean elevation than Group III ($p = <0.001$) and Group II ($p = 0.01$). Mean slope steepness and slope aspect did not vary significantly among the groups ($p = 0.21$ and $p = 0.30$, respectively).

Species composition and tree establishment following fire

All five dominant tree species were present in Group I, but not in Groups II and III (Fig. 3.4). Group I was dominated by *P. menziesii* and *L. occidentalis*, which were each present at 91% of sites in this group. Both high severity fire groups (II and III) were void of *L. occidentalis*, but only Group III lacked *P. menziesii*. In comparison, Group III was dominated by *A. lasiocarpa*, which was present at 80% of sites in this group. This variation in composition was also reflected in species importance values among fire groups. The importance value for *P. menziesii* was significantly greater in Group I than III, while the opposite was true for *A. lasiocarpa* ($p < 0.05$ for both species). The importance value for *L. occidentalis* was significantly greater in Group I than Groups II and III ($p < 0.05$).

Establishment lags following fires of different severity differed among species (Fig. 3.5). Most *L. occidentalis* (79%) established as cohorts within 20 years of a fire, primarily following low-to-moderate- severity fires (Fig. 3.5). While the majority of *P. contorta* (89%) trees also established as cohorts, these were largely in response to high-severity fires. The oldest trees at each site consisted of a distinct *Pinus contorta* cohort, providing evidence that these stands originated after a high-severity fire. A smaller proportion of *P. menziesii* (54%)

established as cohorts, with 64% of trees establishing in association with low-to-moderate severity fires. Of the two most shade-tolerant species, *P. engelmannii* (67%) formed more cohorts than *A. lasiocarpa* (17%). Establishment of both species was continuous at all sites, except site 20, and they had long establishment lags (≥150 years) following high-severity fires.

Structural attributes among fire classes

Most sites had abundant live trees in the 5-24.9 cm and 25-44.9 cm dbh classes (Fig. 3.6a and b). Among the fire history groups, 100% of sites in Group I included large trees (dbh ≥ 45 cm), but the percentage of sites with large trees in Groups II and III were lower (50% and 80%, respectively). However, there was no significant difference in the density of large trees among the fire history groups ($p = 0.20$). Similarly, Group I had the largest trees and greatest variability in tree size, but these attributes did not vary significantly among fire history groups ($p = 0.27$ and 0.12, respectively) (Fig. 3.6c and d).

Snags with dbh >25cm that were not killed by mountain pine beetle were more common at sites in Group III (80%) than in Groups I (27%) and II (50%) (Fig. 3.6). Snag density was significantly greater for Group III than I ($p = 0.043$); however, snag density in Group II was not significantly different from either of these groups ($p > 0.05$ for both pairwise tests).

Species composition relative to topographic variables and structural attributes

Based on the PCA of species importance values, sites were associated with three compositional gradients, which were associated with elevation followed by aspect (Fig. 3.7, Table. 3.5). Principal Component 1 (PC1) was negatively correlated with importance values of *P. menziesii* but positively correlated with those of *A. lasiocarpa* ($r > |0.5|$). PC1 scores were strongly correlated to elevation ($r = 0.77$). Snags were also positively correlated to PC1 ($r = 0.64$), suggesting they were more commonly associated with *A. lasiocarpa* than *P. menziesii*. *P. contorta* was the dominant species associated with Principal Component 2 (PC2, $r = -0.81$); however, axis scores were not significantly correlated to any topographical or structural attributes. Principal Component 3 (PC3) was primarily associated with *L. occidentalis* ($r = -0.79$). PC3 scores were positively correlated with slope aspect ($r = 0.71$), suggesting *L. occidentalis* was more common on sites with cool northeast aspects (low values) rather than warm southwest aspects (high values) (Table. 3.5). No structural attributes were associated with PC3.

3.5 Discussion

What is the prevalence of mixed-severity fire across the landscape?

Historical tree-ring reconstructions provided evidence of mixed- and high-severity fire histories at individual sites in the JGC watersheds over the past 400 years. At the site level, 11 of 20 sites were classified as mixed-severity (Group I), with fire scars, veteran trees and evidence of even-aged cohorts. The remaining nine sites were classified as high-severity, given the presence of a single even-aged cohort and absence of fire scars or veteran trees. Four sites had burned within the last 150 years, representing young forests (i.e., recently initiated by a high-severity fire) (Group II). Five of the high-severity sites had not burned in 274–350 years and thus, were designated as old forests (Group III).

The term “mixed-severity” can describe either: (1) the variation in severity of different fires at a site through time; or (2) the variation in severity within an individual burn (Perry et al. 2011, Heyerdahl et al. 2012). I found evidence of both effects. Fires of differing severity burned at each individual mixed-severity site over time (Fig. 3.2a). Most reconstructed fires (73%) were documented solely by fire scars, indicating many fires were of low-to-moderate severity. The remaining 27% of fires were severe enough to create conditions suitable for cohort establishment at these same sites (Heyerdahl et al. 2012). Mixed-severity effects were also observed within the boundary of a single fire, as found in historic (e.g., Brown et al. 1999; 2008) and contemporary fires (e.g., Lentile et al. 2006; Thompson and Spies 2010) in many other regions. The widespread and well-documented fire of 1910 created high-severity effects in some locations (Da Silva 2009), but also illustrated heterogenous results within an individual fire. Evidence of the 1910 fire was found at 12 study sites, but only initiated a stand at one site. At 11 other sites, it initiated seven cohorts and scarred trees. A quarter of the 215 sampled trees that survived the 1910 fire were fire-intolerant species, found at six different sites. The survival of fire-intolerant species suggests the 1910 fire was either of very low severity or quite patchy *among* and even *within* the sites, as seen elsewhere (Taylor and Skinner 2003).

Although the absence of scars alone does not constitute evidence of high-severity fire (Falk et al. 2011; Schoennagel et al. 2011), three additional lines of evidence support the inference of high-severity for nine sites: (1) veteran trees were absent or sparse (Table 3.3); (2) cohorts were dominated by *P. contorta*, which largely regenerates from serotinous cones in this region (Antos and Parish 2002); and (3) cohorts established over a short period, generally ≤ 15 years (Antos and Parish 2002; Amoroso et al. 2011). Detecting even-aged cohorts is difficult as time-since-last-fire increases (Heyerdahl et al. 2012). This is a shortcoming of using static-age structures for inferences about stand dynamics (Johnson et al. 1994). To overcome this limitation, I used two sets of criteria for defining cohorts that established before and after 1800 (after Ehle and Baker 2003) (described in Chapter 2 methods). These criteria effectively identified *P. contorta* cohorts. When cohorts co-occurred with a fire scar, tree pith dates lagged the fire scars by 9 ± 6 years (mean ± standard deviation). In comparison, when cohorts occurred in absence of fire scars, the establishment window was 6 ± 5 years before 1800 and 4 ± 3 years after 1800. Had *P. contorta* cohorts established following a mountain pine beetle infestation, trees would have established over longer periods (Axelson et al. 2009).

How is fire severity influenced by topography?

Spatial patterns of fire severity were primarily controlled by elevation, with a mixed-severity regime dominant at lower-elevations and transitioning into a high-severity regime at upper-elevations. While the watershed as a whole could be classified as a "mixed-severity regime", given the occurrence of mixed- and high-severity sites, the separation of fire severity along an elevation gradient suggests two fire regimes are present. This pattern is consistent with other mountain landscapes (e.g., Taylor and Skinner 2003; Sherriff and Veblen 2006; Margolis and Balmat 2009), although this research identified mixed-severity sites at much higher elevations (600+ m higher) than previously for this landscape (B.C. Ministry of Forests 1995). Lower elevations with drier fuels relative to higher elevations, helps promote spread of low intensity surface fires (Schoennagel et al. 2004). A contrasting dynamic occurs in higher elevation forests where cooler temperatures and deep, long lasting snow packs increase fuel moisture and shorten the fire season (Schoennagel et al. 2004).

Fuels of higher moisture content are effective barriers to spread of low-severity fires and generally require extreme drought conditions to burn (Schoennagel et al. 2004; Gedalof et al. 2005). In contrast to the impact of elevation, aspect and slope did not show clear associations with fire history groups in the JGC, potentially due to a lack of terrain complexity (Taylor and Skinner 2003).

How do fire, topography and their interaction influence forest composition and structure?

Spatial patterns of fire severity were controlled by interactions between elevation and other environmental gradients (e.g., species composition) that influence fuel characteristics and their flammability (Heyerdahl et al. 2001, Schoennagel et al. 2004). Elevation strongly influences the spatial distribution of tree species in mountain landscapes, yet this pattern is also affected by fire severity. Fire severity influenced species composition, the timing of establishment for different species (Fig. 3.5) and age-structure characteristics (Tab. 3.4, Fig. 3.5). For instance, most (84%) *Larix occidentalis* established following a low-to-moderate severity fire and generally as part of a cohort (65%). In contrast, most *Pinus contorta* (89%) established as a cohort, primarily following high-severity fires. Age-structures of shade-tolerant species (*Abies lasiocarpa* and *Picea engelmanii*) were also influenced by fire severity (Fig. 3.5). While the duration of a fire is short relative to the lifespan of trees, the patterns and structures generated can persist for long periods of time.

Mixed-severity sites were all found below 1700 m.a.s.l., yet the presence of *Larix occidentalis* snags and logs are potential indicators of past low-to-moderate severity fire at high-severity sites above this elevation. *L. occidentalis* is extremely shade-intolerant, requiring abundant light to regenerate. It also requires a concurrent seed source within the period of time when it has a competitive advantage over other shade-tolerant species (e.g., within two years post-disturbance) (Schmidt et al. 1976). While it has been suggested that high-severity fires can occur within *L. occidentalis* forests (Barrett et al. 1991; Davis 1980; Arno 1976), *L. occidentalis* likely requires partial disturbances with veteran trees (as a seed source) for population survival at a stand-level. In this study, *L. occidentalis* was only found at mixed-severity sites (Group I) (including stumps, seedlings and saplings). Yet, evidence of a charred *L. occidentalis* log with a single fire scar was found at a high-severity site at 1871 m.a.s.l. (Site 17), providing two indicators of past surface fire at this site. At another high-severity site (Site 16, 1746 m.a.s.l.), three 275-year old *L. occidentalis* died during the 1910 fire, but no larch regenerated during the 20th century. While the 1910 fire had high-severity effects at this site, the presence of *L. occidentalis* snags may signify the fire regime included some lower severity fires in the past at this elevation.

The transition from a mixed- to a predominantly high-severity regime was not abrupt along an elevational boundary. Sites of all three fire history groups were present between *c.* 1380 and 1700 m.a.s.l. where the spatiotemporal patterns of fire likely constitutes a shifting mosaic. The fire history in some stands may be self-reinforcing due to factors that influence fire spread and intensity (e.g., stand structure, species composition and topography). In addition, previous disturbances (i.e., ecological memory) (Perry 1995; Peterson 2002) can influence the characteristics and outcomes of subsequent fires. For instance, the presence of *L. occidentalis* was associated with cool northeast aspects, which may make these stands more prone to mixed- rather than high-severity fire. In other stands, there may be no self-reinforcing mechanisms as determinants of fire severity are related to weather and stochastic processes (Lertzman et al. 1998; Perry et al. 2011). This may be the case with some high-severity sites as no topographical features explained the spatial distribution of young *versus* old forests subject to high-severity fires.

Distinct forest structural attributes have been associated with some mixed-severity fire regimes (e.g., Spies et al. 2006), but this relationship was not as evident in the JGC watersheds, with the exception of snag densities. Structural attributes (size classes, largest tree, tree-size variability) were more variable within than among fire history groups. High variability in mixed-severity systems could be explained by stands burned by both low- and high-severity fires (Perry et al. 2011). Snags were more abundant at older high-severity sites and consisted primarily of dead standing *Abies lasiocarpa*. These snags were likely generated by processes other than fire given the importance of partial disturbances and autogenic processes in spruce-fir (*Picea engelmanni – A. lasiocarpa*) forests, the dominant forest type in Group III (high-severity, old forest) (Antos and Parish 2002).

Few snags were found at mixed-severity sites; however, this finding may understate the influence of low-to-moderate severity fires on snag creation for three reasons. First, snag densities vary with time-since-last-fire (TSLF) and are greatest: (1) immediately following a fire (e.g.,TSLF <20), or (2) after long periods without fire (e.g., TSLF >250 years, Group III) (Morrison and Raphael 1993; Lehmkuhl et al. 2003; Hutto 2006). As such, I would expect low snag densities at mixed-severity sites as they have not recorded a fire in 56 to 99 years, although densities may have once been similar as those found in older high-severity forests (Hutto 2006). Second, live green trees with decay were not included as snags in the analysis, despite their functional similarity for many wildlife species (Bull et al. 1997). Many *L. occidentalis* could have been classified as 'live green trees with decay' as heartwood decay (*Fomitopsis officinalis)* is very common in old veteran *L. occidentalis* and often accelerated by fire damage (Bull et al. 1997; Gyug et al. 2009). Third, selective harvesting (c. 1910-1950) of >300 year old *L. occidentalis* at many sites removed potential snag recruits (Fig. 3.2a *versus* 3.2b/c).

Strong agreement between two approaches used to infer severity

Understanding the agreement among methods used to infer historic fire severity is key to understanding the questions posed in this research. In this study, variation in age-structure indices among fire history groups corroborated with site-level inferences based on scars and cohorts. Mixed-severity sites (Group I) were uneven-aged with trees present in numerous but discontinuous age classes. Uneven-age structures are driven by low-to-moderate or patchy fires which leave large thick-barked trees and some fire-intolerant trees intact (Beaty and Taylor 2007). Discontinuous age-structures result from fires periodically initiating tree establishment and eliminating susceptible trees (Beaty and Taylor 2007; Brown et al. 2008) or non-fire factors that decrease germination, establishment or recruitment (e.g., unfavorable climate, lack of viable seeds or limited growing space) (Johnson and Fryer 1989; Amoroso et al. 2011). Similarly, old stands initiated by a high-severity fire (Group III) were uneven-aged but had more continuous tree establishment and survival since the last fire (> 250 years ago). The oldest trees at each site were part of a post-fire *Pinus contorta* cohort, but subsequent periods of low establishment and survival resulted from stand development (e.g., crown closure, competition and self thinning) (Johnson and Fryer 1989; Antos and Parish 2002). Shade-tolerant *Abies lasiocarpa* and *Picea engelmannii* (Fig. 3.5) subsequently established in the understory over long periods, sometimes forming new cohorts (e.g., Site 15)(Antos and Parish 2002). In contrast, sites that established after recent high-severity fires

(Group II) had the simplest age-structure generally represented by one discrete continuous cohort (Taylor and Skinner 2003).

3.6 Conclusions

Mixed-severity fire regimes were prevalent in the Joseph and Gold Creek watersheds yet, low-to-moderate severity fires have largely been eliminated due to 20th century fire exclusion in mixed-severity systems (Daniels et al. 2011). Here, only four localized fires have burned in the last 100 years (Da Silva 2009). The low-to-moderate severity surface fires, which were historically frequent, are presently missing from these forests (Da Silva 2009), and are an important component of a mixed-severity regime. Silvicultural practices have largely replaced fire as the dominant disturbance agent in the JGC watersheds.

Management and research in mixed-severity systems must focus on the ecological processes affected by exclusion of surface fires. Ecological restoration, fuel-management plans and silvicultural approaches need to emulate (or reintroduce) the low-to-moderate severity component of the mixed-severity fire regime. Towards this goal, this study outlines four important findings for mixed-severity fire regimes:

1. Fire regimes differentiate along elevational gradients with gradual boundaries that may change over time;
2. Veteran trees and *L. occidentalis* are associated with mixed-severity regimes;
3. Many differences in forest structure between mixed- and high-severity regimes are subtle given the large amounts of variability within individual stands;
4. Inferences of fire severity based on the presence of fire scars on live trees and even-aged cohorts match inferences based solely on age-structure.

Variation in fire severity contributes to stand- and landscape-level heterogeneity in several ways (Perry et al. 2011). Infrequent high-severity fires can have a dominant influence on tree establishment by creating stands with homogenous age-structures (Taylor and Skinner 2003; Turner et al. 2003). Low-to-moderate severity fires can reduce fuel densities (Bekker and Taylor 2010), facilitate snag recruitment (Hutto 2006) and increase the heterogeneity of understory plant communities (Peterson and Reich 2008). Individual trees and patches surviving a mixed-severity fire create biological legacies such as seed sources for regeneration (Spies et al. 2006). By producing a diversity of developmental pathways for individual stands, mixed-severity regimes also result in heterogeneity at the broader landscape scale. As a result, landscapes dominated by mixed-severity regimes potentially support a greater diversity of plants and animals than either low- or high-severity systems (Perry et al. 2011). Thus, an improved understanding of the assumptions underlying fire severity research in general, and mixed-severity regimes in particular, is an imperative for restoration, conservation, forest management and fire hazard mitigation in the urban-wildland interface.

Table 3.1 Site-level attributes derived for 20 research sites and methods used to obtain them.

Site-level attributes	Method
Fire & disturbance history	
Fire scars (number and years)	Up to 10 fire-scarred trees sampled in a 1-ha plot and dated using standard techniques.
Cohorts (number and years)	Identified using a sliding window to detect 20-year periods when: ≥ 30 trees ha^{-1} (pre-1800) or ≥ 50 trees ha^{-1} (post-1800) established.
Time-since-last-fire	Calculated as the number of years between the most recent fire scar or cohort date and the year of sampling (i.e., 2009).
Veteran density (ha^{-1})	Identified as canopy trees (dbh ≥25 cm) that pre-dated and survived at least one fire.
Establishment Lags*	Determined for each individual tree following low-to-moderate or high-severity fire.
Age structure	Derived two indices, evenness and continuity, by counting the number of 15-year age classes occupied and unoccupied by trees.
Topography	
Elevation (m.a.s.l.)	Measured in field using GPS.
Slope (degrees)	Measured in field using clinometers.
Aspect (warm-to-cool index)	Measured in field using compass. Slope aspect was converted into a linear index representing warm (0 = southwest) to cool (180 = northeast) aspects.
Composition	
Species importance values	Calculated using the sum of relative basal area and relative density (index value between 0−200).
Structural attributes	
Tree-size variability	Estimated using coefficient of variation of basal area.
Largest tree	Based on maximum diameter at breast height (dbh).
Small tree density (5-24.9 cm dbh)	Estimated using live tree tallies in 10 x 10 subcanopy plot.
Medium (25-44.9 cm dbh) & Large (≥45 cm dbh) tree density	Derived from *N*-tree canopy plot used to select the 30 live trees, snags or stumps (dbh ≥ 25 cm dbh) closest to plot center.
Snag density (≥25 cm dbh)	Derived from *N*-tree canopy plot. Estimate included all dead-standing trees, except for mountain pine beetle (MPB) killed trees. No green trees with decay were included.

* Establishment lags are the only attribute estimated at a tree-level rather than at a site-level.

Table 3.2 Stand-level evidence used to differentiate the fire history of 20 research sites.

	Fire History Group		
Criteria	I Mixed Severity Young Forest	II High Severity Young Forest	III High Severity Old Forest
Fire scars on living trees	present	absent	absent
Time-since-last-fire	<150 yrs	<150 yrs	≥150 yrs

Table 3.3 Summary of fire history evidence, forest composition and tree densities for 20 sites classified by fire history group.

Fire history group	Site	Elev. (m)	Number of fires		Last fire (year)	Time since last fire (years)	Relative composition (species* importance values)					Density (individuals per ha)			
			cohort	scar			Psme	Laoc	Pico	Pien	Abla	Live trees 5–24.9 cm	Live trees ≥ 25 cm	Veterans ≥ 25 cm	Snags† ≥ 25 cm
I. Mixed Severity, Young Forest															
	1	1168	1	6	1922	87	77	123	0	0	0	2400	94	1876	15
	2	1259	2	6	1910	99	82	93	25	0	0	800	114	91	0
	3	1287	1	15	1941	68	194	0	0	0	0	0	298	258	0
	4	1320	1	5	1910	99	169	31	0	0	0	100	361	265	0
	5	1330	4	6	1910	99	13	145	40	3	0	3400	278	179	10
	6	1352	1	4	1931	78	26	95	79	0	0	1100	177	973	0
	7	1366	3	9	1910	99	117	83	0	0	0	4000	159	138	0
	8	1386	1	3	1924	85	0	115	81	4	0	500	282	910	0
	10	1420	3	7	1953	56	2	29	16	133	19	1600	399	1928	0
	12	1581	2	2	1910	99	1	10	187	0	2	5400	61	32	12
	13	1693	2	3	1910	99	1	75	6	56	62	1800	189	57	0
II. High Severity, Young Forest															
	9	1417	1	0	1935	74	196	0	0	0	0	300	148	5	0
	14	1693	1	0	1865	144	0	0	75	105	14	300	616	0	21
	16	1746	1	0	1910	99	0	0	109	61	30	1400	233	0	8
	17	1871	1	1‡	1905	104	0	0	181	0	19	2167	543	0	0
III. High Severity, Old Forest															
	11	1570	1	0	1721	288	0	0	0	20	180	4700	227	0	79
	15	1739	2	0	1721	288	0	0	6	24	171	900	463	0	51
	18	1872	1	0	1659	350	0	0	5	34	162	1100	188	0	89
	19	1905	1	0	1661	348	0	0	32	53	116	2000	194	0	40
	20	1981	1	0	1735	274	0	0	200	0	0	300	270	0	0

*Species abbreviations are Psme = *Pseudotsuga menziesii*, Laoc = *Larix occidentalis*, Pico = *Pinus contorta*, Pien = *Picea engelmannii*, and Abla = *Abies lasiocarpa*.

† Snags do not include *P. contorta* trees snags resulting from a regional mountain pine beetle infestation.

‡ This scar was found on a charred dead tree.

Table 3.4 Comparison of age structure indices (eveness and continuity) among fire history groups. Different superscripts indicate significant differences in the number of 15-year age classes among groups ($\alpha = 0.05$).

Fire history group	n	Evenness (Number of occupied age classes per site)		Continuity (Number of unoccupied age classes per site)	
		Mean (SE)	Range	Mean (SE)	Range
I. Mixed severity, young forest	11	7.7 (0.7)^{a}	3−11	9.5 (1.5)^{a}	0−16
II. High severity, young forest	4	3.5 (1.0)^{b}	2−6	0.5 (0.5)^{b}	0−2
III. High severity, old forest	5	9.4 (1.9)^{a}	3−14	3.4 (1.6)^{b}	0−7

Table 3.5 Correlations between PCA axis scores (PC 1, 2, 3) and species importance values, site topography and forest structure. PCA axis scores were derived from an ordination of species importance values for the 20 sites. Only correlation values >|0.5|(*asterix) are significant.

	PC1	PC2	PC3
Importance Values			
Pseudotsuga menziesii	-0.53*	0.39	0.45
Larix occidentalis	-0.44	0.10	-0.79*
Pinus contorta	0.12	-0.81*	0.26
Picea engelmanii	0.49	0.12	-0.32
Abies lasiocarpa	0.52*	0.42	0.1
Topography			
Elevation	0.77*	-0.28	0.32
Slope steepness	0.33	0.00	0.3
Slope aspect	-0.16	0.04	0.71*
Structure			
Tree-size variability	0.12	-0.18	-0.31
Largest tree	0.17	-0.30	-0.28
Small (5−24.9 cm dbh) ha^{-1}	0.10	0.15	-0.18
Med. (25−44.9 cm dbh) ha^{-1}	0.41	0.19	0.09
Large (≥45 cm dbh) ha^{-1}	0.15	-0.45	-0.37
Snag (≥25 cm dbh) ha^{-1}	0.64*	-0.44	0.08

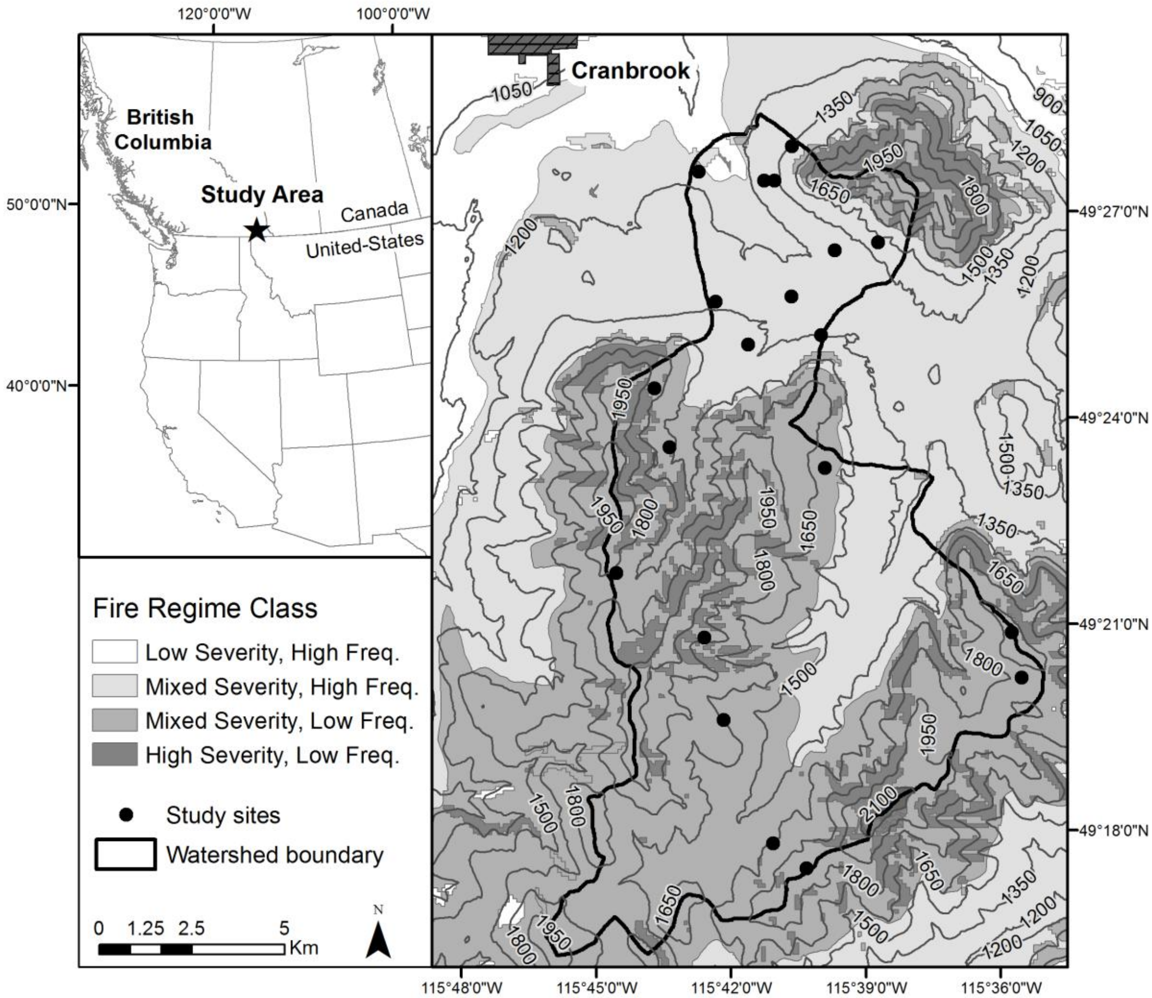


Fig. 3.1 Location of the 20 study sites in the Joseph and Gold Creek watersheds in southeastern British Columbia, Canada. These watersheds provide the drinking water supply for the City of Cranbrook, located northwest of the study area. Ten sites were randomly located in each of the two fire regime classes (Mixed Severity, High Frequency and Mixed Severity, Low Frequency) that account for 87% of the study area.

a. Fire History Group I (Mixed severity, young forest)

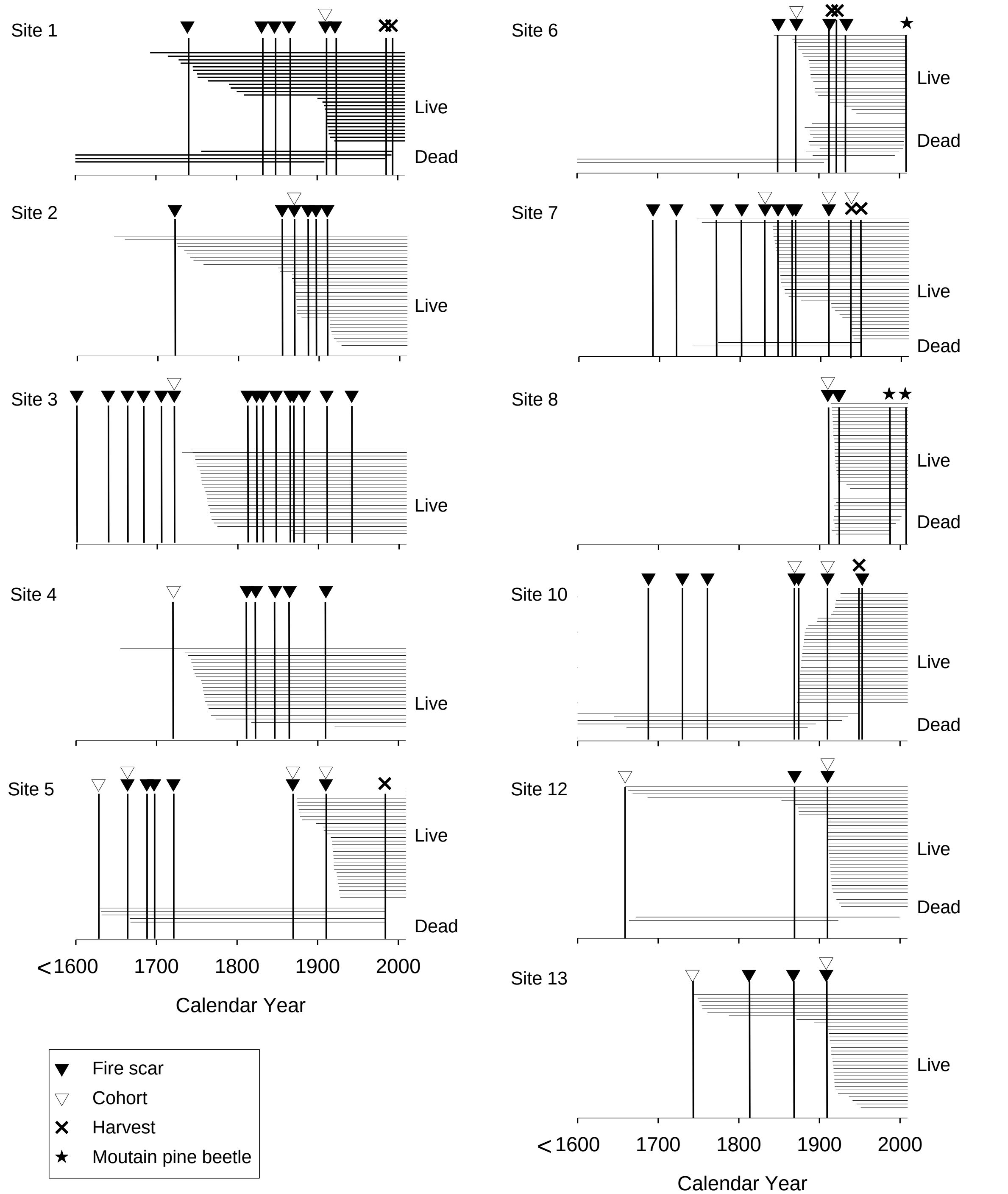
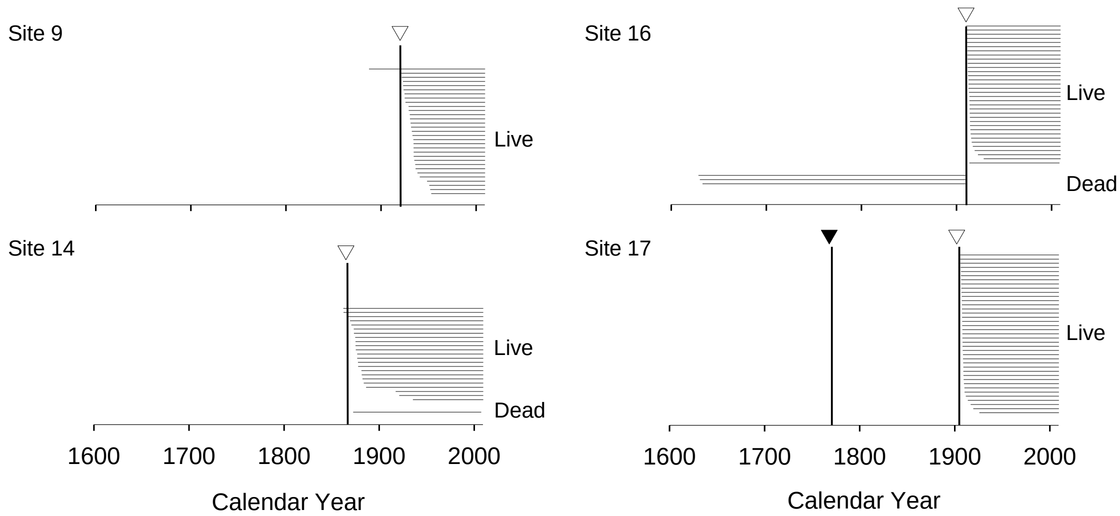


Fig. 3.2 Fire demography diagrams for the 20 sites grouped by fire history (I, II and III). Each diagram represents all live and dead trees (horizontal lines) crossdated at a single site. Fire scar dates are denoted by black triangles and even-aged cohorts by white triangles. Other disturbance agents that resulted in tree deaths include 20th century harvesting (cross) and mountain pine-beetle (MB, star).

b. Fire History Group II (High severity, young forest)



c. Fire History Group III (High severity, old forest)

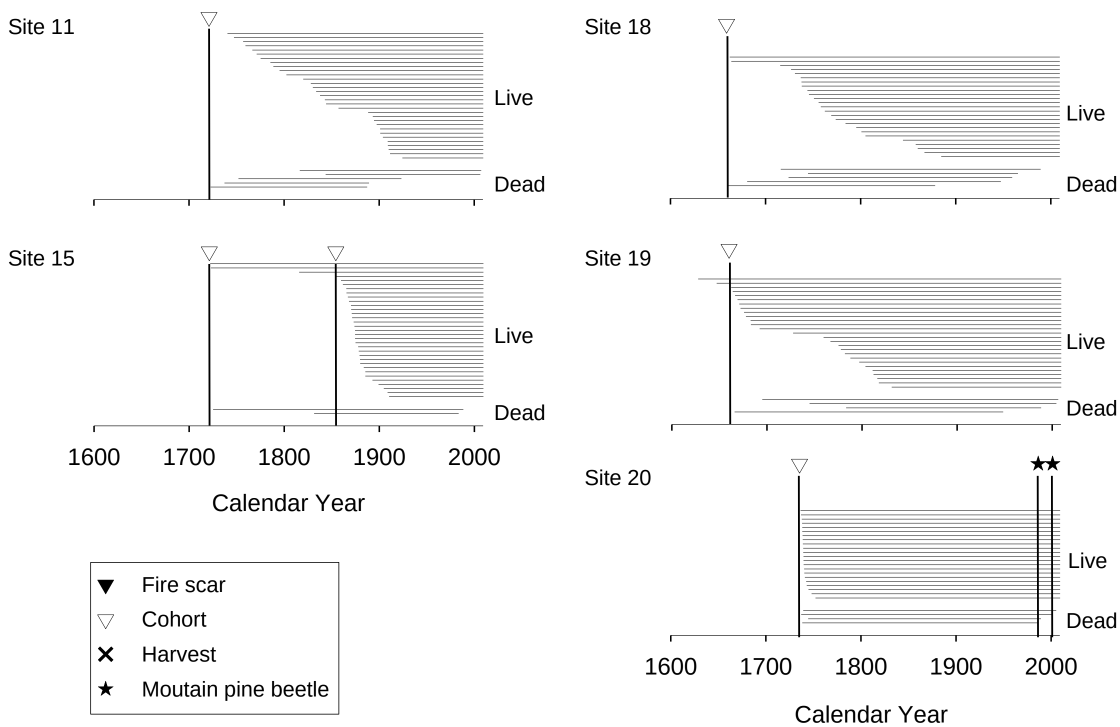


Fig. 3.2 Fire demography diagrams for the 20 sites grouped by fire history (I, II and III). Each diagram represents all live and dead trees (horizontal lines) crossdated at a single site. Fire scar dates are denoted by black triangles and even-aged cohorts by white triangles. Other disturbance agents that resulted in tree deaths include 20th century harvesting (cross) and mountain pine-beetle (MPB, star).

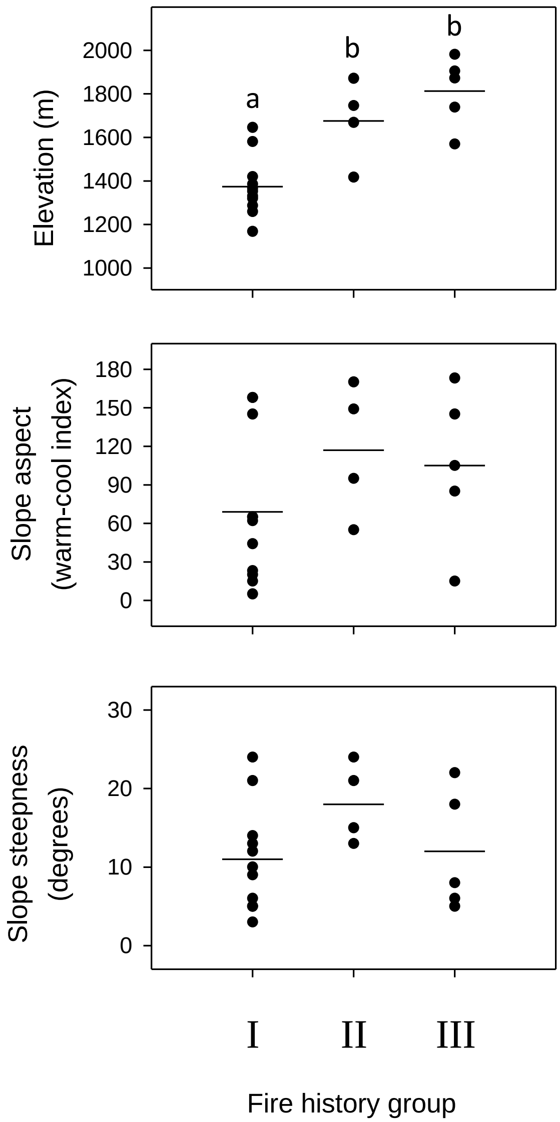


Fig. 3.3 Comparison of environmental characteristics among fire history groups. Slope aspect was transformed into a linear value ranging from 0 (warm = 225°) to 180 (cool = 45°). Horizontal lines represent means. Different superscripts indicate significant differences in means among fire history classes ($\alpha = 0.05$).

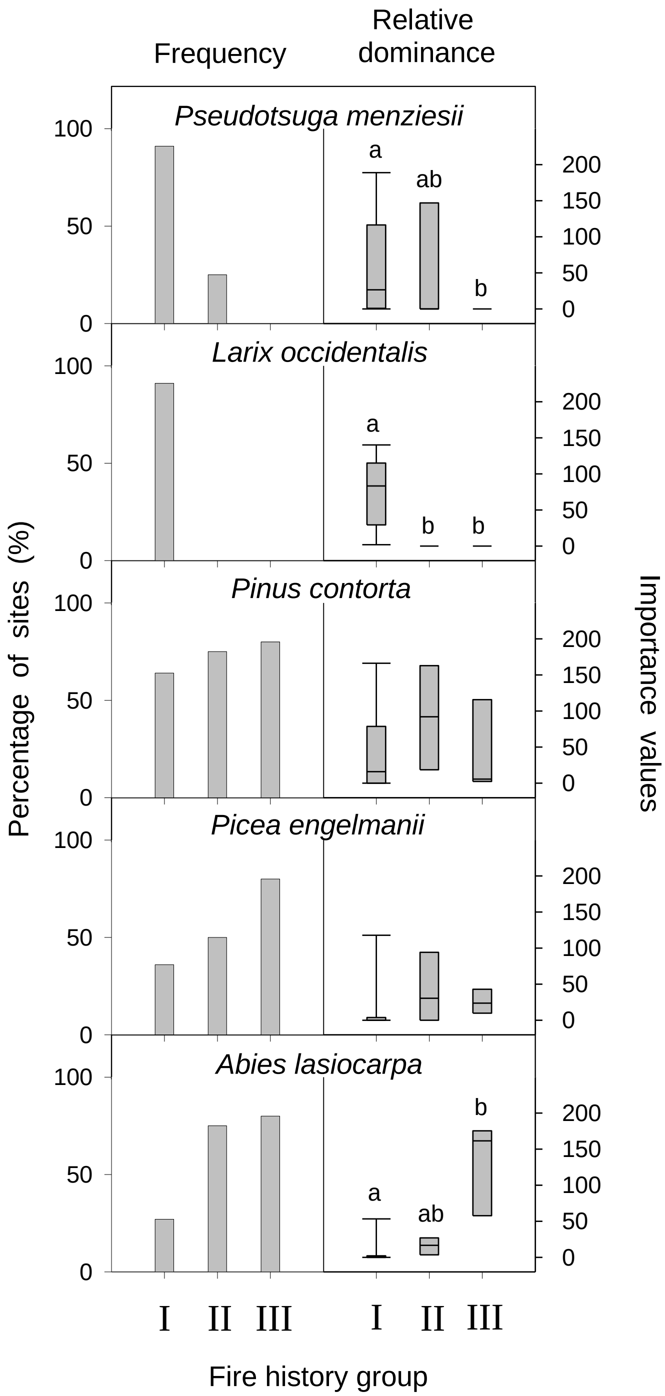


Fig. 3.4 Comparison among fire history groups of relative dominance (left) and frequency (right) of the five dominant tree species. All live trees with dbh ≥ 5 cm are included. Importance values (IV) were calculated for each site as the sum of relative density and relative basal area and range from 0 to 200. In each box plot, the black horizontal line represents the median and box boundaries are the 25th and 75th percentiles; and, bars are the 10th and 90th percentiles. Different superscripts indicate significant differences in medians among fire history groups (α = 0.05).

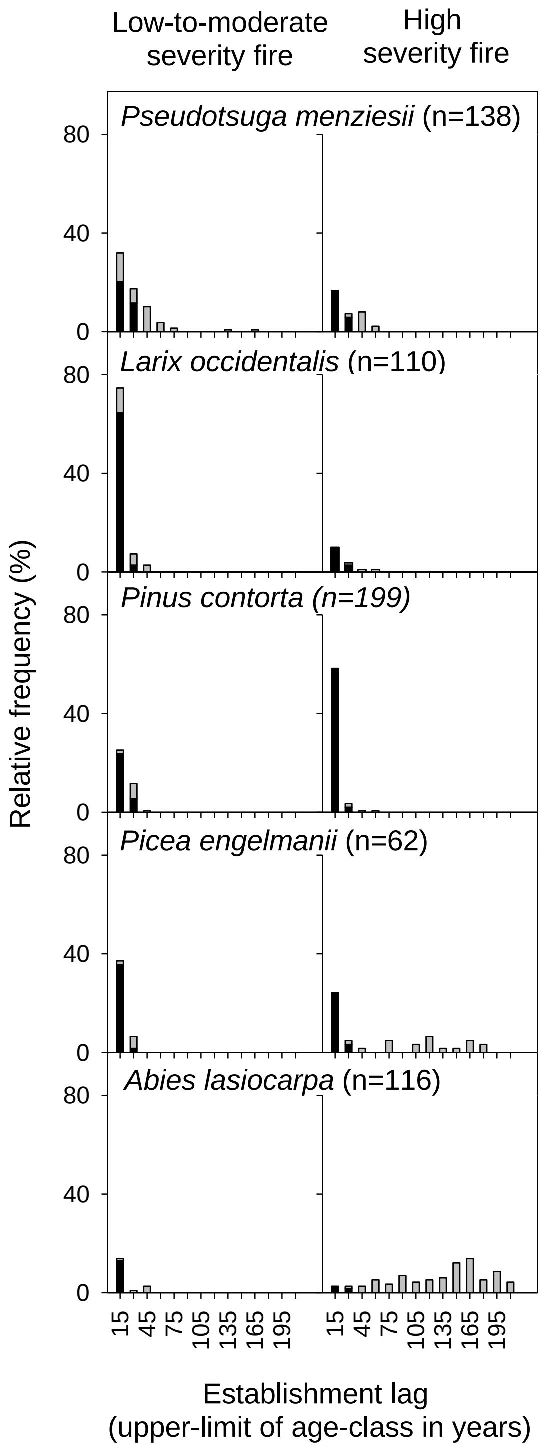


Fig. 3.5 Lags in tree establishment following low-to-moderate-severity fires (left) and high-severity fires (right) fires for trees forming cohorts (black) versus trees not in cohorts (grey). Lags following low-to-moderate-severity fires were calculated as the difference between fire-scar years and estimated pith date of individual trees and for high-severity fires they were calculated as the difference between the estimated pith dates of the oldest tree in the cohort and individual trees.

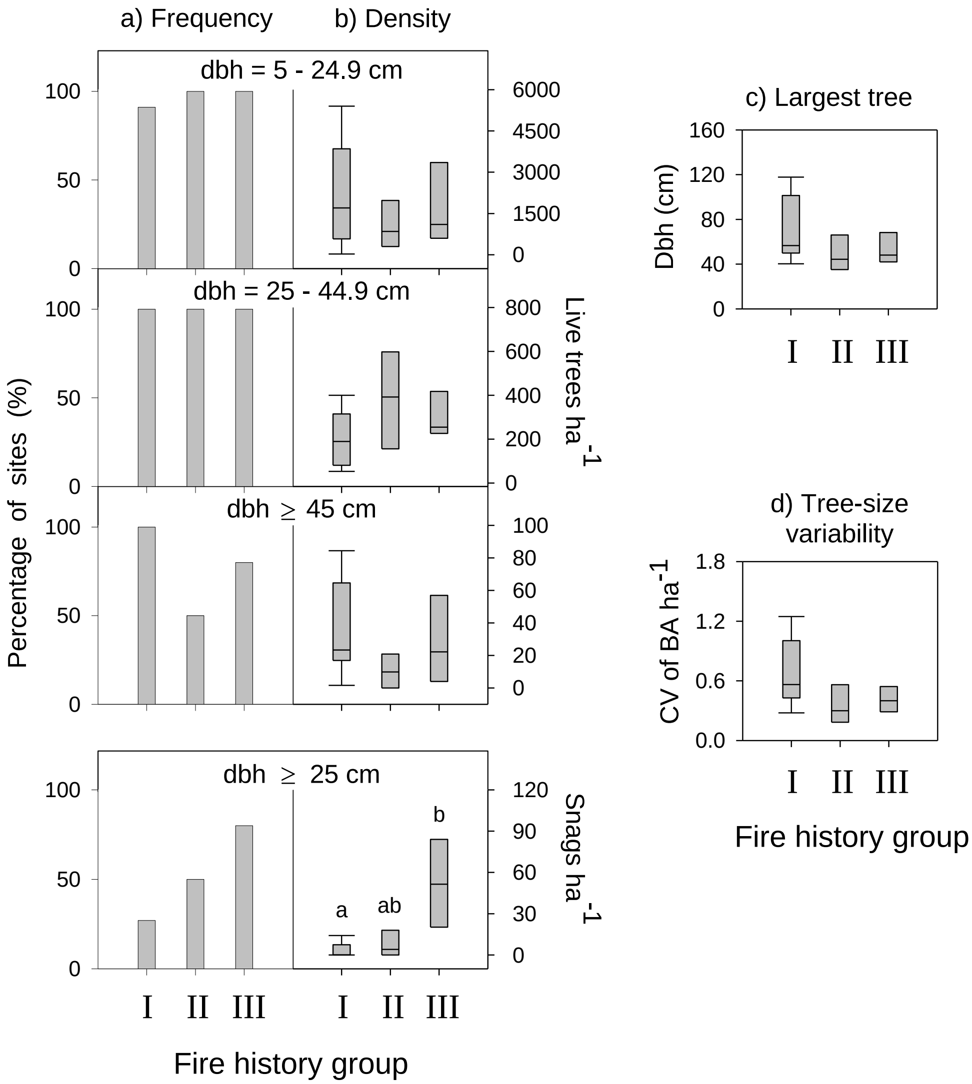


Fig. 3.6 Comparison of structural attributes among fire history groups. Attributes are : (a) frenquency and (b) density per site of live trees (three size classes) and snags, (c) largest tree at each site, and (d) tree-size variability based on the coefficient of variation (CV) of basal area (BA) ha^{-1} of trees with dbh >25 cm. In each box plot, the black horizontal line represents the median and box boundaries are the 25^{th} and 75^{th} percentiles; and, bars are the 10^{th} and 90^{th} percentiles. Different superscripts indicate significant differences in medians among fire history groups ($\alpha = 0.05$).

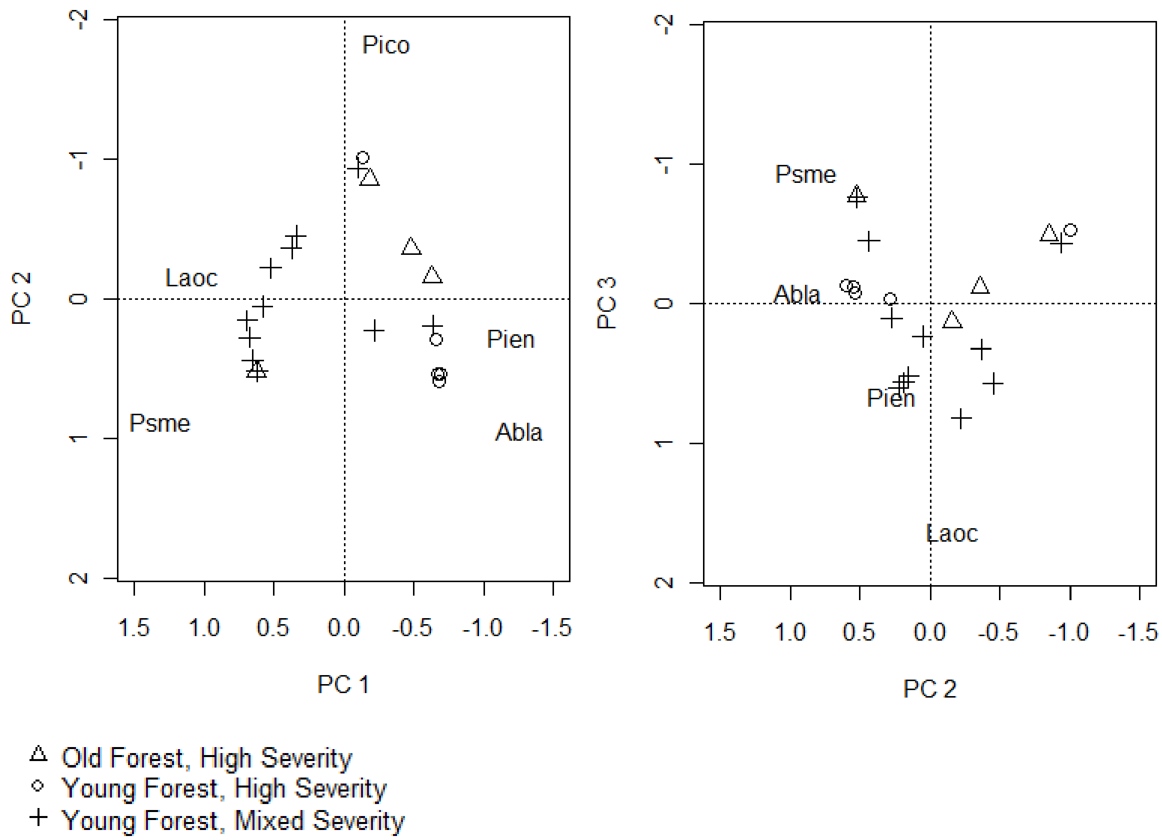


Fig. 3.7 Principal components analysis of forest composition using species' importance values for 20 sites. Sites are grouped by fire history. The cumulative proportion of variance explained by all three components is 85% (PC1 = 38%, PC2 = 28%, PC3 = 19%). Species abbreviations are Psme = *Pseudotsuga menziesii*, Laoc = *Larix occidentalis*, Pico = *Pinus contorta*, Pien = *Picea engelmannii*, and Abla = Abies lasiocarpa.

Chapter 4: Conclusions

4.1 Summary and contributions to current research

This study contributes to the growing body of knowledge addressing the spatial extent of mixed-severity fire regimes in British Columbia (B.C.). In particular, the importance of mixed-severity fire regimes in mountain forests is poorly quantified in southeastern B.C. Until recently, fire regimes in this region were considered predominantly high-severity, with low-severity regimes confined to some valley bottoms (B.C. Ministry of Forests 1995). For example, high-severity fire regimes have been estimated to dominate Rocky Mountain National Parks (e.g., Van Wagner et al. 2006; Johnson et al. 1990; Masters 1990). However, mixed-severity regimes were likely common in forests located upslope from valley bottom forests (Gray et al. 2002). They were also historically prevalent in lower-elevation ponderosa pine (*Pinus ponderosa*) and Douglas-fir (*Pseudotsuga menziesii*) forests (Klenner et al. 2008; Heyerdahl et al. 2012) and mid-elevation mixed-conifer forests in southeastern regions (Cochrane 2007; Da Silva 2009; Nesbitt 2010; Greene 2011).

My research suggests existing fire regime classification systems misrepresent mixed-severity regimes at mid-elevations. Few studies have used field-based evidence to validate fire history classification schemes (for another example see Swetnam and Brown 2010), particularly in mixed-severity systems. Although currently used to guide B.C. forest management, the NDT system under-represents mixed-severity regimes. Fire frequency was also underestimated by the NDT classification. In contrast, the HNFR classification (developed for assessing fire risk in B.C.) over-represented mixed-severity regimes and under-represented fire regimes with longer fire return intervals. These classification errors can be linked to underlying assumptions about disturbances and limitations of fire history research methods (Chapter 2).

More studies based on spatially representative stands and that address fires of a range of severities are needed, particularly for mid-elevations. For instance, fire histories of some forest types are less certain than others due to lack of studies (e.g., Wong and Iverson 2004; Swetnam and Brown 2010). Chapter 2 shows how scaling-up or extrapolating (Miller et al. 2004) from reference studies can oversimplify fire history at the watershed- and landscape-scales. Using case studies for estimating fire history attributes to parts of the landscape where no fire history knowledge exists is a valid approach, but must be used with caution (Wong et al. 2003). Knowledge drawn from case studies should be used as a hypothesis until area specific information becomes available (Veblen 2003).

Elevation is a strong driver of fire behaviour, but elevational boundaries between fire regime types are not abrupt (Chapter 3). Other factors are likely driving this pattern, including interactions between top-down (e.g., climate variation) and bottom-up (e.g., fire weather, fuel composition and configuration) (Heyerdahl et al. 2002, 2007; Falk et al. 2007, 2011). Unlike other studies (e.g., Heyerdahl et al. 2001; Fulé et al. 2003), I did not find aspect or slope steepness to be significant contributors to spatial variation of severity. However, species composition, fire weather and stochasticity were likely important factors.

Fire-adapted *Larix occidentalis* (western larch) may be an indicator of mixed-severity fire (Chapter 3). Few fire history studies have been published for *L. occidentalis* forest types (except Barrett et al. 1991). Two fire regimes have been suggested for *L. occidentalis* forests: 1) mixed-severity regimes on dry sites with variable fire frequencies (e.g., 25-75 years), 2) high-severity regimes on moist sites after longer fire intervals (e.g., 120-350 years)(Barrett et al. 1991, Davis 1980, Arno 1976). However, it is unlikely *L. occidentalis* would re-establish after high-severity fires. *L. occidentalis* requires survivor trees or a seed source in close proximity in order for it to re-establish after a fire (Schmidt et al. 1976). In contrast to other species, *L. occidentalis* must also establish within two years of a fire due to its extreme lack of shade-tolerance (Schmidt et al. 1976). Mixed-severity systems have a high edge to interior ratio with lots of small patches (Perry et al. 2011), two landscape attributes that would aid the proliferation of this species over time.

4.2 Linking conclusions to management and conservation

Tackling fire-regime classification errors in British Columbia

Existing fire-regime classification errors could be corrected by reevaluating our understanding of fire regimes relative to Biogeoclimatic Ecosystem Classification (BEC) units and adjusting elevational boundaries of each fire regime type. Most mixed-severity sites were located in the Montane Spruce (MS) zone, which is situated in a mid-elevational band between the low-elevation Interior Douglas-fir (IDF) and high-elevation Engelmann Spruce-Subalpine Fir (ESSF) zones (Meidinger and Pojar 1991). For instance, 88% of sites (7 of 8) in the MS zone but only 20% of sites (2 of 10) in the ESSF zone had mixed-severity fire histories. Similarly, a recent study estimated that at least 25% of the MS zone in the southern Rocky Mountain Trench of B.C. was historically characterized by low-to-moderate severity fires (Cochrane 2007). Therefore, extending the elevational boundary of the NDT4 class (low severity, high frequency fires) from the upper limit of the IDF zone to the upper limit of the MS zone would rectify many errors of the NDT classification system. The extent of mixed-severity fires in the higher elevation ESSF zone is unclear (Stuart-Smith and Hendry 1998; Gray et al. 2002), and may be best categorized as a predominantly high-severity class (NDT3 and HNFR5). Two sites located in the low-elevation IDF zone were also classified as mixed-severity, a finding that is supported by other studies in this forest type (Heyerdahl et al. 2007, 2012; Klenner et al. 2008).

Adopting non-equilibrium forest successional models

Traditional equilibrium successional paradigms do not apply in mixed-severity systems. Forest management needs to expand beyond this equilibrium paradigm to improve biodiversity and conservation planning. The notion that predictable stages of stand development occur until a climax forest develops has been used in high-severity systems. For example, this underlying assumption validates the use of seral-stage management for landcape-level planning in B.C. (B.C. Ministry of Forests 1995). Seral-stage management uses the equilibrium successional model to maintain landscape heterogeneity by ensuring forests in different developmental stages are present and their relative abundance is consistent with the frequency of stand-replacing fires (B.C. Ministry of Forests 1995). It assumes stand development stages following high-severity fires (e.g. stand initiation, stem-exclusion, understory-reinitiation, old-growth stages in Oliver and Larson 1990) can be tracked using stand-age. However, stand-age information in B.C. forest inventories (e.g., based on age of dominant cohort, or time since last fire) are poor descriptors of forest dynamics in mixed-severity systems.

Strengths and limitations of present work

The strengths of this research include (1) approaches designed to be scientifically robust at both site- and landscape-levels and (2) the use of multiple lines of evidence to strengthen reconstructions of historic fire regimes (e.g., Margolis et al. 2007). Both these factors are of particular importance when studying mixed-severity regimes (Agee 2005; Heyerdahl et al. 2012). Selecting spatially representative sites is also crucial for extrapolations across the study watersheds. Here, the stratified-random site selection improved upon the targeted sampling methods often used in fire-scar research (e.g., Arno and Sneck 1977). Site-level fire history interpretations used multiple criteria (cohorts, fire scars and time since last fire), which were corroborated using veteran trees and two indices of age-structures (Chapter 3). The focus on mid-elevation forests is also of particular importance given the lack of research in this part of the landscape.

Although my findings demonstrate that mixed-severity regimes historically occurred in the Montane Spruce (MS) zone, my findings cannot be directly extrapolated to determine the proportion of the MS characterized by this regime type. Spatial inferences about broader landscape units (e.g., across the entire Montane Spruce BEC zone) would require a random selection of sites across the whole MS zone, or the portion with fire-tolerant trees (e.g., Cochrane 2007)

Forest structural differences among fire history groups could be improved using temporally specific assessments (e.g., chronosequences, Agee and Huff 1987; Stuart-Smith et al. 2006) involving sampling more recently burned sites (e.g., time-since-last-fire < 50 years). Most mixed-severity sites within this study burned over 100 years ago, despite historical records of more frequent fires (Da Silva 2009). Alternatively, comparisons of stand structural attributes immediately after a mixed- and high-severity fire (e.g., 0 to 5 years after fire) would strengthen inferences about direct effects of fire on snag dynamics and size structure.

4.3 Future recommendations for fire history research

Addressing spatial limitations of dendrochronology with aerial photographs

The heterogeneous nature of fire likely impedes our ability to scale up site-level estimates of fire severity to the entire landscape. Stand structural attributes (e.g., overstory and understory canopy cover, species composition, stem densities) derived from aerial photograph interpretation (API) have been used as proxies for fire severity (Hessburg et al. 2007; Taylor and Skinner 1998; Beaty and Taylor 2008). Thus, API offers opportunities to address questions over broader spatial scales, whereas dendrochronology provides detailed temporal information (Hessburg et al. 2007). In particular, historic aerial photographs (*c.* 1930's) could be used to reconstruct fire history prior to intensive forest management. Few studies have compared fire severity estimates derived from dendrochronology with those based on aerial photographs (Swetnam and Brown 2010; Swetnam et al. 2011). Understanding the concordance among dendrochronology and API approaches would improve our ability to map fire severity and other fire regime attributes (e.g., fire perimeters, Swetnam et al. 2011).

Understanding the fire exclusion problem and solutions in mixed-severity systems

The inherent variability of mixed-severity fire regimes creates ambiguity about what represents a “natural” forest condition (Klenner et al. 2008, Halofsky et al. 2011). This variability also makes it difficult to determine whether fuel mitigation and restoration targets are needed and whether they could be effective in reducing the risk of fire to human property. Understanding the extent of fuel-buildup and what mitigation or restoration approaches are needed to maintain ecosystem services and protect communities living at the urban-wildland interface is a management imperative.

Detecting the role of climate in tree establishment

Climate variability can not only influence fire occurrence, but also tree establishment. Cool wet climates can prevent fires from burning but also promote tree establishment (Brown and Wu 2005). With climate driven tree establishment, I would expect to see cohort establish synchronously across the watersheds. In contrast, cohorts that establish after a fire would be more site-specific except after large regional fires also be driven by climate (e.g., 1910 fire). Assessing the role of climate in tree establishment could be accomplished by comparing cohort establishment dates with a tree-ring reconstruction of the Palmer Drought Severity Index (Cook et al. 2004).

Effect of 20th century forest management and fire suppression on western larch forests

Large western larch (*Larix occidentalis*) trees could be used as an indicator of changes on the landscape. Loss of large veteran trees is an important management concern (Ellison et al. 2005) given their importance to key ecological functions. In western Canada and the United-States, losses of large veteran larch are a particular concern for various primary cavity nesters (Bull et al. 1997; Gyug et al. 2009). Harvesting of large diameter western larch was common in the early- and mid- 20th century (Hessburg et al. 2000) and little attention was given to larch regeneration (Carlson et al. 1995). Despite an abundance of larch stumps throughout its range (Hessburg et al. 2000), it is unclear how present larch forest structures differ from the past. Active suppression of wildfires may also contribute to changes in forest structure and composition. Western larch's shade-intolerance and dependence on mixed-severity fires for regeneration (Chapter 3) may have decreased in abundance due to fire exclusion (Fiedler and Lloyd 1995; Carlson et al. 1995; Thompson 1995). Forest in-growth (due to fire exclusion) may also precipitate the death of veteran larch through competition for moisture and nutrients (Carlson et al. 1995).

4.4 Conclusion

By incorporating more rigorous fire history methods, I have improved our knowledge of a poorly understood fire regime types. Ecological heterogeneity created by mixed-severity regimes potentially influences decisions related to conservation, silviculture, wildfire and fuel mitigation. Mixed-severity regimes may also change dramatically with a warming climate (Westerling et al. 2006). Thus, understanding controls on fire severity will help ecologists and managers foresee potential changes in forests and fire regimes, and guide more informed management of the important services derived from forests.

References

Agee, J.K., and Huff, M.H. 1987. Fuel succession in a western hemlock/Douglas-fir forest. Canadian Journal of Forest Research **17**(7): 697-704.

Agee, J.K. 1993. Fire ecology of pacific northwest forests. Island Press, Washington, D.C.

Agee, J. 2005. The complex nature of mixed severity fire regimes. *In* Mixed severity fire regimes: ecology and management. *Edited by* Lagene, L., J. Zelnik, S. Cadwallader, and B. Hughes. Washington State University Coop Extension Service/ The Association for Fire Ecology, Spokane, Washington. pp. 1-10.

Aho, K. 2011. asbio: A collection of statistical tools for biologists. R package version 0.3-39. Available from: http://CRAN.R-project.org/package=asbio (accessed January 7, 2013).

Amoroso, M.M., Daniels, L.D., Bataineh, M., and Andison, D.W. 2011. Evidence of mixed-severity fires in the foothills of the Rocky Mountains of west-central Alberta, Canada. Forest Ecology and Management **262**(12): 2240–2249.

Antos, J.A., and Parish, R. 2002. Dynamics of an old-growth, fire-initiated, subalpine forest in southern interior British Columbia: tree size, age, and spatial structure. Canadian Journal of Forest Research **32**(11): 1935–1946.

Arno, S.F. 1976. The historical role of fire on the Bitterroot National Forest USDA Forest Service, Intermountain Forest and Range Experiment Station, Research Paper INT-187.

Arno, S.F., and Sneck, K.M. 1977. A method for determining fire history in coniferous forests of the mountain west. *In*, Ogden, UT. p. 28.

Axelson, J.N., Alfaro, R.I., and 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.

B.C. Forest Analysis and Inventory. 2012. Biogeoclimatic ecosystem classification (BEC) map. GeoBC Data Discovery Services. Victoria, B.C. Available from http://www.for.gov.bc.ca/hre/becweb/resources/maps/GISdataDownload.html (accessed , May 10, 2010).

B.C. Ministry of Forests. 1995. Biodiversity Guidebook – Forest Practices Code of British Columbia. Victoria, B.C. Available from http://www.for.gov.bc.ca/tasb/legsregs/fpc/fpcguide/biodiv/biotoc.htm (accessed September 15, 2012).

Barrett, S.W. 1981. Relationship of indian-caused fires to the ecology of western Montana forests. M.S.c. Thesis. University of Montana, Missoula, MT.

Barrett, S.W., and Arno, S.F. 1982. Indian fires as an ecological influence in the northern Rockies. Journal of Forestry **80**(10): 647-651.

Barrett, S.W., Arno, S.F., and Key, C.H. 1991. Fire regimes of western larch - lodgepole pine forests in Glacier National Park, Montana. Canadian Journal of Forest Research **21**(12): 1711–1720.

Beaty, R.M., and Taylor, A.H. 2007. Fire disturbance and forest structure in old-growth mixed conifer forests in the northern Sierra Nevada, California. Journal of Vegetation Science **18**(6): 879.

Beaty, R.M., and Taylor, A.H. 2008. Fire history and the structure and dynamics of a mixed conifer forest landscape in the northern Sierra Nevada, Lake Tahoe Basin, California, USA. Forest Ecology and Management **255**(3): 707-719.

Beck, J. 1984. Fire history and range enhancement study for the Cayuse Creek area. Report submitted to Selkirk College, Castlegar, B.C.

Bergeron, Y., Leduc, A., Harvey, B., and Gauthier, S. 2002. Natural fire regime: a guide for sustainable management of the Canadian boreal forest. Silva Fennica **36**(1): 81–95.

Blackwell, B.A., Gray, R.W., Green, R.N., Feigl, F., Berry, T., Ohlson, D., and Hawkes, B. 2003. Developing a coarse scale approach to the assessment of forest fuel conditions in southern British Columbia. Report submitted to Forest Innovation Investment. Victoria, B.C.

Bowman, D.M., Balch, J.K., Artaxo, P., Bond, W.J., Carlson, J.M., Cochrane, M.A., D'Antonio, C.M., DeFries, R.S., Doyle, J.C., and Harrison, S.P. 2009. Fire in the Earth system. Science **324**(5926): 481-484.

Brown, P.M., and Swetnam, T.W. 1994. A cross-dated fire history from coast redwood near Redwood National Park, California. Canadian Journal of Forest Research **24**(1): 21–31.

Brown, P.M., Kaufmann, M.R., and Shepperd, W.D. 1999. Long-term, landscape patterns of past fire events in a montane ponderosa pine forest of central Colorado. Landscape Ecology **14**(6): 513-532.

Brown, P.M., and Wu, R. 2005. Climate and disturbance forcing of episodic tree recruitment in a southwestern ponderosa pine landscape. Ecology **86**(11): 3030-3038.

Brown, P.M. 2006. Climate effects on fire regimes and tree recruitment in Black Hills ponderosa pine forests. Ecology **87**(10): 2500–2510.

Brown, P.M., Wienk, C.L., and Symstad, A.J. 2008. Fire and forest history at Mount Rushmore. Ecological Applications **18**(8): 1984-1999.

Bull, E.L., Parks, C.G., and Torgersen, T.R. 1997. Trees and logs important to wildlife in the Interior Columbia River Basin. USDA For. Serv. Gen. Tech. Rept. PNW-391. Pacific Northwest Forest and Range Experimentation Station, Portland, OR.

Carlson, C.E., Byler, J.W., and Dewey, J.E. 1995.Western larch: pest tolerant conifer of the Rocky Mountains. In: Schmidt, W.C., McDonald, K.J. (Comps.), Ecology and management of larix forest: a look ahead. USDA, Forest Service Intermountain Research Station General Technical Report, GTR-INT-319.

Clague, J.J. 1975. Late Quaternary sediments and geomorphic history of the southern Rocky Mountain Trench, British Columbia. Canadian Journal of Earth Sciences **12**(4): 595-605.

Cochrane, J.D. 2007. Characteristics of historical forest fires in complex mixed-conifer forests of southeastern British Columbia. M.S.c. Thesis. Department of Geography, University of British Columbia, Vancouver, B.C.

Collins, B.M., and Stephens, S.L. 2010. Stand-replacing patches within a ‘mixed severity’ fire regime: quantitative characterization using recent fires in a long-established natural fire area. Landscape Ecology **25**: 927-939.

Cook, E.R., Woodhouse, C.A., Eakin, C.M., Meko, D.M., and Stahle, D.W. 2004. Long-term aridity changes in the western United States. Science **306**(5698): 1015-1018.

Cissel, J.H., Swanson, F.J., and Weisberg, P.J. 1999. Landscape management using historical fire regimes: Blue River, Oregon. Ecological Applications **9**(4): 1217–1231.

Da Silva, E. 2009. Wildfire history and its relationship with top-down and bottom-up controls in the Joseph and Gold Creek watersheds, Kootenay Mountains, British Columbia. M.S.c. Thesis, Department of Geography, University of Guelph, Guelph, O.N.

Daniels, L.D., Maertens, T.B., Stan, A.B., McCloskey, S.P.J., Cochrane J.D. and Gray, R.W. 2011. Direct and indirect impacts of climate change on forests: three case studies from British Columbia. Canadian Journal of Plant Pathology **33**(2): 108–116

Davis, K.M. 1980. Fire history of a western larch/Douglas-fir forest type in northwestern Montana. *In* Proceedings of the fire history workshop, Tuscon, AZ, October 20-24, 1984. *Edited by* M.A. Stokes and J.H. Dieterich. U.S. Dep. Agric. For. Serv. Gen. Tech. Rep. RM-81., Fort Collins, CO.

Demaerschalk, J.P., and Omule, S.A. 1982. Estimating breast height diameters from stump measurements in British Columbia. The Forestry Chronicle **58**(3): 143-145.

Donovan, G.H., and Brown, T.C. 2007. Be careful what you wish for: the legacy of Smokey Bear. Frontiers in Ecology and the Environment **5**(2): 73-79.

Duncan, R. 1989. An evaluation of errors in tree age estimates based on increment cores in Kahikatea (*Dacrycarpus dacrydioides*). New Zealand Natural Sciences **16**: 31–37.

Ehle, D., and Baker, W. 2003. Disturbance and stand dynamics in ponderosa pine forests in Rocky Mountain National Park, USA. Ecological Monographs **73**(4): 543–566.

Ellison, A.M., Bank, M.S., Clinton, B.D., Colburn, E.A., Elliott, K., Ford, C.R., Foster, D.R., Kloeppel, B.D., Knoepp, J.D., and Lovett, G.M. 2005. Loss of foundation species: consequences for the structure and dynamics of forested ecosystems. Frontiers in Ecology and the Environment **3**(9): 479-486.

Environment Canada. 2011. Canadian climate normals 1971–2000. Meteorological Service of Canada. Available from http://www.climate.weatheroffice.gc.ca/climate_normals/index_e.html. [accessed February 12, 2011].

Falk, D.A., Miller, C., McKenzie, D., and Black, A.E. 2007. Cross-scale analysis of fire regimes. Ecosystems **10**(5): 809-823.

Falk, D.A., Heyerdahl, E.K., Brown, P.M., Farris, C., Fulé, P.Z., McKenzie, D., Swetnam, T.W., Taylor, A.H., and Van Horne, M.L. 2011. Multi-scale controls of historical forest-fire regimes: new insights from fire-scar networks. Frontiers in Ecology and the Environment **9**(8): 446–454.

Fiedler, C.E., Lloyd, D.A., 1995. Autecology and synecology of western larch. In: Schmidt, W.C, McDonald, K.J. (Comps.), Ecology and management of larix forest: a look ahead. USDA, Forest Service Intermountain Research Station General Technical Report, GTR-INT-319.

Flannigan, M.D., Logan, K.A., Amiro, B.D., Skinner, W.R., and Stocks, B.J. 2005. Future Area Burned in Canada. Climatic Change **72**(1-2): 1-16.

Franklin, J.F., Spies, T.A., Pelt, R.V., Carey, A.B., Thornburgh, D.A., Berg, D.R., Lindenmayer, D.B., Harmon, M.E., Keeton, W.S., Shaw, D.C., Bible, K., and Chen, J. 2002. Disturbances and structural development of natural forest ecosystems with silvicultural implications, using Douglas-fir forests as an example. Forest Ecology and Management (1-3)**155**: 399–423.

Fulé, P.Z., Heinlein, T.A., Covington, W.W., and Moore, M.M. 2003. Assessing fire regimes on Grand Canyon landscapes with fire-scar and fire-record data. International Journal of Wildland Fire **12**(2): 129-145.

Gedalof, Z., Peterson, D., and Mantua, N. 2005. Atmospheric, climatic, and ecological controls on extreme wildfire years in the northwestern United States. Ecological Applications **15**: 154-174.

Greene, G.A. 2011. Historical fire regime of the darkwoods: quantifying the past to plan for the future. University of British Columbia, Department of Geography, Vancouver, B.C. p. 115.

Gray, R.W., Andrew, B., Blackwell, B.A., Needoba, and A., Steele. 2002. The Effect of physiography and topography on fire regimes and forest communities. Report submitted to Habitat Conservation Trust Fund. Victoria, B.C.

Grissino-Mayer, H.D. 2001. Evaluating crossdating accuracy: a manual and tutorial for the computer program COFECHA. Tree-Ring Research **57**(1): 205–221.

Gyug, L.W., Steeger, C., and Ohanjanian, I. 2009. Characteristics and densities of Williamson's Sapsucker nest trees in British Columbia. Canadian Journal of Forest Research **39**: 2319-2331.

Halofsky, J.E., Donato, D.C., Hibbs, D.E., Campbell, J.L., Cannon, M.D., Fontaine, J.B., Thompson, J.R., Anthony, R.G., Bormann, B.T., Kayes, L.J., Law, B.E., Peterson, D.L., and Spies, T.A. 2011. Mixed-severity fire regimes: lessons and hypotheses from the Klamath-Siskiyou Ecoregion. Ecosphere **2**(4): art40.

Hessburg, P.F., Smith, B.G., Salter, R.B., Ottmar, R.D., and Alvarado, E. 2000. Recent changes (1930s-1990s) in spatial patterns of interior northwest forests, USA. Forest Ecology and Management **136**(1–3): 53-83.

Hessburg, P.F., Salter, R.B., and James, K.M. 2007. Re-examining fire severity relations in pre-management era mixed conifer forests: inferences from landscape patterns of forest structure. Landscape Ecology **22**: 5–24.

Heyerdahl, E.K., Brubaker, L.B., and Agee, J.K. 2001. Spatial controls of historical fire regimes: a multiscale example from the interior west, USA. Ecology **82**(3): 660-678.

Heyerdahl, E., Brubaker, L., and Agee, J. 2002. Annual and decadal climate forcing of historical fire regimes in the interior Pacific Northwest, USA. The Holocene **5**: 597-604.

Heyerdahl, E.K., Lertzman, K., and Karpuk, S. 2007. Local-scale controls of a low-severity fire regime (1750-1950), southern British Columbia, Canada. Ecoscience **14**: 40-47.

Heyerdahl, E., Lertzman, K., and Wong, C.M. 2012. Mixed-severity fire regimes in dry forests of southern interior British Columbia, Canada. Canadian Journal of Forest Research **42**(1): 88–98.

Hirsch, K.G. and P. Fuglem, P., Technical Coordinators. 2006. Canadian wildland fire strategy: background syntheses, analyses, and perspectives. Canadian Council of Forest Ministers, Natural Resources Canada, Canadian Forest Service, Northern Forestry Centre, Edmonton, Alberta,113 p.

Hutto, R.L. 2006. Toward meaningful snag-management guidelines for postfire salvage logging in North American conifer forests. Conservation biology **20**(4): 984-993.

Johnson, E., and Fryer, G. 1989. Population dynamics in lodgepole pine-Engelmann spruce forests. Ecology **70**(5): 1335-1345.

Johnson, E., Fryer, G., and Heathcott, M. 1990. The influence of man and climate on frequency of fire in the interior wet belt forest, British Columbia. Journal of Ecology **78**(2): 403–412.

Johnson, E., Miyanishi, K., and Kleb, H. 1994. The hazards of interpretation of static age structures as shown by stand reconstructions in a Pinus contorta--Picea engelmannii forest. Journal of Ecology: 923-931.

Johnson, E., and Gutsell, S. 1994. Fire frequency models, methods and interpretations. Advances in Ecological Research **25**: 239–287.

Keane, R.E., Hessburg, P.F., Landres, P.B., and Swanson, F.J. 2009. The use of historical range and variability (HRV) in landscape management. Forest Ecology and Management **258**: 1025-1037.

Klenner, W., Walton, R., Arsenault, A., and Kremsater, L. 2008. Dry forests in the Southern Interior of British Columbia: Historic disturbances and implications for restoration and management. Forest Ecology and Management **256**(10): 1711-1722.

Kutner, M.H., Nachtsheim, C.J., Neter, J., and Li, W. 2005. Applied linear statistical models, 5th edition. McGraw-Hill, Boston, MA.

Landres, P., Morgan, P., and Swanson, F. 1999. Overview of the use of natural variability concepts in managing ecological systems. Ecological Applications **9**(4): 1179-1188.

Lehmkuhl, J.F., Everett, R.L., Schellhaas, R., Ohlson, P., Keenum, D., Riesterer, H., and Spurbeck, D. 2003. Cavities in snags along a wildfire chronosequence in eastern Washington. The Journal of wildlife management **67**: 219-228.

Lentile, L.B., Smith, F.W., and Shepperd, W.D. 2006. Influence of topography and forest structure on patterns of mixed severity fire in ponderosa pine forests of the South Dakota Black Hills, USA. International Journal of Wildland Fire **15**(4): 557-566.

Lertzman, K., Fall, J., and Dorner, B. 1998. Three kinds of heterogeneity in fire regimes: at the crossroads of fire history and landscape ecology. Northwest Science **72**: 4–23.

Lynch, T.B., and Rusydi, R. 1999. Distance sampling for forest inventory in Indonesian teak plantations. Forest Ecology and Management **113**(2): 215-221.

Mah, S. 2000. Relationship between vital attributes of Ktunaxa plants and natural disturbance regimes in southeastern British Columbia. M.S.c. Thesis. The University of British Columbia. Faculty of Forestry. Vancouver, B.C.

Margolis, E.Q., Swetnam, T.W., and Allen, C.D. 2007. A stand-replacing fire history in upper montane forests of the southern Rocky Mountains. Canadian Journal of Forest Research **37**(11): 2227-2241.

Margolis, E.Q., and Balmat, J. 2009. Fire history and fire–climate relationships along a fire regime gradient in the Santa Fe Municipal Watershed, NM, USA. Forest Ecology and Management **358**(11): 2416-2430.

Masters, A. 1990. Changes in forest fire frequency in Kootenay National Park, Canadian Rockies. Canadian Journal of Botany **68**(8): 1763–1767.

Meidinger, D., and Pojar, J. 1991. Ecosystems of British Columbia. Spec. Rep. Ser. No. 6. B.C. Ministry of Forests, Victoria, B.C., Available from http://www.for.gov.bc.ca/hfd/pubs/Docs/Srs/Srs06.htm (accessed April 2, 2011).

Morgan, P., Hardy, C.C., Swetnam, T.W., Rollins, M.G., and Long, D.G. 2001. Mapping fire regimes across time and space: understanding coarse and fine-scale fine patterns. International Journal of Wildland Fire **10**(4): 529–342.

Morrison, M.L., and Raphael, M.G. 1993. Modeling the dynamics of snags. Ecological Applications: 322-330.

Nesbitt, J.H. 2010. Quantifying forest fire variability using tree rings, Nelson, British Columbia, 1700-present. M.S.c. Thesis. University of British Columbia, Department of Geography, Vancouver, B.C.

Oksanen, J.F., Blanchet, G., Kindt, R., Legendre, P., O'Hara, R.B., Simpson, G.L., Solymos, P., Stevens, M.H.H., and Wagner, H. 2011. vegan: Community Ecology Package. R package version 1.17-6. Available from : http://CRAN.R-project.org/package=vegan (accessed March 5, 2013).

Oliver, C.D., and Larson, B.C. 1990. Forest stand dynamics. McGraw-Hill, Inc.

Peres-Neto, P.R., Jackson, D.A., and Somers, K.M. 2003. Giving meaningful interpretation to ordination axes: assessing loading significance in principal component analysis. Ecology **84**(9): 2347-2363.

Perry, D.A. 1995. Self-organizing systems across scales. Trends in Ecology & Evolution **10**(6): 241-244.

Perry, D.A., Hessburg, P.F., Skinner, C.N., Spies, T.A., Stephens, S.L., Taylor, A.H., Franklin, J.F., McComb, B., and Riegel, G. 2011. The ecology of mixed severity fire regimes in Washington, Oregon, and Northern California. Forest Ecology and Management **262**(5): 703–717.

Peterson, D.W., and Reich, P.B. 2008. Fire frequency and tree canopy structure influence plant species diversity in a forest-grassland ecotone. Plant Ecology **194**(1): 5-16.

Peterson, G.D. 2002. Contagious disturbance, ecological memory, and the emergence of landscape pattern. Ecosystems **5**(4): 329-338.

Richman, M.B. 1988. A cautionary note concerning a commonly applied eigenanalysis procedure. Tellus **40B**: 50-58.

Schellhaas, R., Camp, A.E., Spurbeck, D., and Keenum, D. 2000. Report to the Colville National Forest on the results of the South Deep watershed fire history research. U.S. Dep. Agric. For. Serv. PNW Res. Sta., Wenatchee For. Sci. Lab., Wenatchee, WA.

Schmidt, K.M., Menakis, J.P., Hardy, C.C., Hann, W.J., and Bunnell, D.L. 2002. Development of coarse-scale spatial data for wildland fire and fuel management. U.S. Dep. Agric. For. Serv. Gen. Tech. rep. RMRS-87. Ogden, UT.

Schmidt, W.C., Shearer, R.C., and Roe, A.L. 1976. Ecology and Silviculture of Western Larch Forests. U.S. Department of Agriculture, Forest Service, Technical Bulletin No. 1520.

Schoennagel, T., Veblen, T., and Romme, W. 2004. The interaction of fire, fuels, and climate across Rocky Mountain forests. BioScience **54**(7): 661–676.

Schoennagel, T., Sherriff, R.L., and Veblen, T.T. 2011. Fire history and tree recruitment in the Colorado Front Range upper montane zone: implications for forest restoration. Ecological Applications **21**(6): 2210-2222.

Schoennagel, T., Veblen, T.T., Negron, J.F., and Smith, J.M. 2012. Effects of mountain pine beetle on fuels and expected fire behavior in lodgepole pine forests, Colorado, USA. PloS one **7**(1): e30002.

Sherriff, R.L., and Veblen, T.T. 2006. Ecological effects of changes in fire regimes in *Pinus ponderosa* ecosystems in the Colorado Front Range. Journal of Vegetation Science **17**(6): 705.

Sherriff, R.L., and Veblen, T.T. 2007. A spatially-explicit reconstruction of historical fire occurrence in the ponderosa pine zone of the Colorado Front Range. Ecosystems **10**(2): 311-323.

Sinton, D.S., Jones, J.A., Ohmann, J.L., and Swanson, F.J. 2000. Windthrow disturbance, forest composition, and structure in the Bull Run basin, Oregon. Ecology **81**(9): 2539-2556.

Soverel, N.O., Coops, N.C., Perrakis, D.D., Daniels, L.D., and Gergel, S.E. 2011. The transferability of a dNBR-derived model to predict burn severity across 10 wildland fires in western Canada. International Journal of Wildland Fire **20**(4): 518-531.

Spies, T.A., Hemstrom, M.A., Youngblood, A., and Hummel, S. 2006. Conserving old-growth forest diversity in disturbance-prone landscapes. Conservation biology **20**(2): 351-362.

Stephens, S.L., and L. W. Ruth. 2005. Federal forest-fire policy in the United States. Ecological Applications **15**(2): 532–542.

Stuart-Smith, K., and Hendry, R. 1998. Residual trees left by fire: final report. British Columbia Ministry of Forests, Enhanced Forest Management Pilot Project Summary. Invermere, BC.

Stuart-Smith, A.K., Hayes, J.P., and Schieck, J. 2006. The influence of wildfire, logging and residual tree density on bird communities in the northern Rocky Mountains. Forest Ecology and Management **231**(1): 1-17.

Swetnam, T.L., and Brown, P.M. 2010. Comparing selected fire regime condition class (FRCC) and LANDFIRE vegetation model results with tree-ring data. International Journal of Wildland Fire **19**(1): 1-13.

Swetnam, T., Falk, D., Hessl, A., and Farris, C. 2011. Reconstructing landscape pattern of historic fires and fire regimes. *In* Landscape ecology of fire. *Edited by* McKenzie D, Miller C, and Falk DA. Springer, Dordrecht, The Netherlands.

Taylor, A.H., and Skinner, C.N. 1998. Fire history and landscape dynamics in a late-successional reserve, Klamath Mountains, California, USA. Forest Ecology and Management **111**(2-3): 285-301.

Taylor, A.H., and Skinner, C.N. 2003. Spatial patterns and controls on historical fire regimes and forest structure in the Klamath Mountains. Ecological Applications **13**(3): 704-719.

Thompson, J.R., and Spies, T.A. 2010. Factors associated with crown damage following recurring mixed-severity wildfires and post-fire management in southwestern Oregon. Landscape Ecology **25**(5): 775-789.

Tompson, C.F. 1995 Management issues of *Larix occidentalis* forests in British Columbia. . In: Schmidt, W.C, McDonald, K.J. (Comps.), Ecology and management of larix forest: a look ahead. USDA, Forest Service Intermountain Research Station General Technical Report, GTR-INT-319.

Turner, M.G., Romme, W.H., and Tinker, D.B. 2003. Surprises and lessons from the 1988 Yellowstone fires. Frontiers in Ecology and the Environment **1**(7): 351-358.

Turner, M. 2010. Disturbance and landscape dynamics in a changing world. Ecology **91**(10): 2833–2849.

Valentine, K., Sprout, P., Baker, T., and Lavkulich, L. 1978. The soil landscapes of British Columbia. Resource Analysis Branch, B.C. Ministry of the Environment.

Van Wagner, C., Finney, M., and Heathcott, M. 2006. Historical fire cycles in the Canadian Rocky Mountain parks. Forest Science **52**(6): 704–717.

Veblen, T.T. 2003. An introduction to key issues in fire regime research for fuels management and ecological restoration. *In* Fire, Fuel Treatments and Ecological Restoration: Conference proceedings. Fort Collins, CO, April 16-18, 2002. *Edited by* P. Omi, and L. Joyce. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Fort Collins, CO. pp. 259-276.

Westerling, A.L., Hidalgo, H.G., Cayan, D.R., and Swetnam, T.W. 2006. Warming and earlier spring increase western U.S. forest wildfire activity. Science **313**(5789): 940-943.

Wimberly, M.C., and Spies, T.A. 2001. Influences of environment and disturbance on forest patterns in coastal Oregon watersheds. Ecology **82**(5): 1443-1459.

Wong, C.M., and Lertzman, K.P. 2001. Errors in estimating tree age: implications for studies of stand dynamics. Canadian Journal of Forest Research **31**(7): 1262–1271.

Wong, C., Dorner, B., Sandmann, H., and Branch, R. 2003. Estimating historical variability of natural disturbances in British Columbia. Ministry of Forests, Ministry of Sustainable Resource Management. Victoria, B.C. Land Management Handbook No. 53.

Wong, C., and Iverson, K. 2004. Range of natural variability: applying the concept to forest management in central British Columbia. BC Journal of Ecosystems and Management **4**(1): 1-14.

Appendices

Appendix A Site numbers cross-referenced with Da Silva 2009

HNFR class	Site number in this thesis	Site number in Da Silva 2009
HNFR 2		
	1	25
	2	29
	3	10
	4	11
	5	9
	6	30
	7	13
	8	22
	9	19
	10	21
HNFR 4		
	11	12
	12	33
	13	8
	14	6
	15	7
	16	18
	17	31
	18	20
	19	16
	20	17