2020

Learning from wilderness fire: restoring landscape scale patterns and processes

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LEARNING FROM WILDERNESS FIRE: RESTORING LANDSCAPE SCALE PATTERNS AND PROCESSES

By

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Thesis

presented in partial fulfillment of the requirements
for the degree of

Master of Science
in Systems Ecology

The University of Montana
Missoula, MT

May 2020

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Wilderness areas, because they are managed to be "untrammeled by man," often offer the best approximation of intact, undisturbed ecological patterns and processes. In the case of wildland fire, this means that wilderness areas often provide the only landscapes where fire has been managed to play an active, ecosystem role. As a result, these wilderness areas offer unique lessons both in terms of wildland fire management as well as the ecological consequences that result from this management approach. For these reasons, an in-depth history of fire management in the wilderness areas of the Northern Rocky Mountains is provided to highlight the lessons learned from these long-running programs where fire has been managed for resource benefit. The four decades of wilderness fire management across three wilderness areas revealed that wildland fire management is most likely to occur when land managers possess a strong commitment to the untrammeled nature of wilderness, when fire management personnel are well versed in long-term fire management strategies and skill sets, and when strong lines of communication are in place, both across administrative boundaries and between land managers and the public. From this history and these lessons learned, recommendations were developed for strengthening wildland fire management for resource benefit across the western U.S., including outside of Congressionally designated wilderness. These recommendations include bolstering the workforce capacity and incentive structure related to fire management for resource benefit, improving communication tactics regarding wildland fire management objectives, and cultivating an ecological fire ethic within land management agencies. Finally, using data collected from mixed-conifer forest plots in the Bob Marshall Wilderness of Northwest Montana, I investigated the forest stand structures that result from an active fire regime. I then identified the pathways to development for the identified stand structures, as well as the drivers of conversion of forest to non-forest structure following fire and the role of fire in creating within-structure class heterogeneity. From these analyses, I produce a data-driven conceptual model of stand structure development under an active fire regime. The results of these studies, taken together, point to the importance of wilderness areas as valuable sources of information on ecosystem processes and patterns, such as wildland fire and forest structure.

ACKNOWLEDGEMENTS

It seemingly takes a village to get a graduate student to her final thesis submission, and I am no exception. For that reason, I would first and foremost like to thank my advisor Andrew Larson for his unwavering support and advice throughout this whole process. Andrew represents a combination of ecological brilliance and high emotional intelligence that can be hard to find, and I am beyond grateful that you were willing to take me on as a graduate student and provide unending support and feedback along the way. I am similarly lucky to have had Travis Belote, Phil Higuera, and Dave Affleck on my committee, as they were always willing to provide invaluable edits or advice. Carol Miller, although not a committee member, was often called upon to play an advisory role, which has thoroughly strengthened the final product. Outside of my committee, however, there are a number of folks associated the University of Montana who provided me with an important support system, including Gretchen McCafferty, Eryn Schneider, Colin Maher, and Abigail Marshall. To my family and friends, both within Missoula and without, thank you for your endless hours of encouragement. The conversations, laughs, drinks, and days climbing I had with you all kept me sane and made what can be an incredibly stressful process infinitely more enjoyable. Finally, I am grateful to the Aldo Leopold Wilderness Research Institute and the Wilderness Institute at UM for their funding and research support.

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INTRODUCTION

The Wilderness Act of 1964 defines designated wilderness as an area that is "untrammeled by man … which is protected and managed so as to preserve its natural conditions and which generally appears to have been affected primarily by the forces of nature" (Wilderness Act 1964). As such, wilderness areas are most often managed to allow, when and where possible, natural processes to play out. For many of the large wilderness areas of the western United States (hereafter, the West), and particularly for the large and relatively remote wilderness areas of the Northern Rockies, this has resulted in vast tracts of wilderness land where fire is managed to play an active ecosystem role.

This approach to fire management in the wilderness areas of the Northern Rockies stands in stark contrast to fire management across the rest of the U.S., where land management agencies tend to suppress the majority of wildland fire ignitions (Wilson et al. 2011). This default to suppression is due to a number of reasons, including the risk that wildland fire poses to human lives and properties as well as the culture of fire suppression that tends to predominate in these land management agencies (Calkin et al. 2013). Wildland fire, however, serves as one of the primary drivers of stand and landscape vegetation structure in the West (Hessburg et al. 2005). Therefore, the result of this suppression-focused management paradigm has been the removal of fire as an active, ecosystem process across the majority of the West, which has resulted in deleterious ecosystem and fire risk effects.

For this reason, wilderness areas offer a unique opportunity to study both management of wildland fire as an ecosystem process, as well as the resulting ecological effects. In my master's thesis I take advantage of this opportunity, and the long history of wilderness fire management in the Northern Rockies in particular, to present research findings intended to guide forest and wildland fire management decisions in the West. Specifically, the results of my research are aimed at improving management of wildland fire for resource benefit, as well as management of forest structure in order to better reflect the patterns created by an active fire regime.

To that end, with chapter 1 of my thesis, I present the history of management of fire as an ecosystem process in wilderness areas of Northern Rockies. Data for this chapter was compiled from a literature review as well as new interviews conducted with wilderness fire managers. I aim to provide land and fire managers with a history that they may build upon and learn from in order to increase use of non-suppression wildland fire management on their own land management units.

Chapter 2 condenses the history detailed in chapter 1 down to the primary lessons learned from nearly four decades of wilderness fire management in the Northern Rockies. The goal of this chapter is to identify and make accessible to a broader audience the core lessons learned from wilderness fire management. Chapter 2 also provides recommendations for applying these lessons learned to fire management nationally in order to increase the extent to which wildland fire is managed for resource benefit.

Finally, Chapter 3 analyzes plot-level forest structure data collected from the Bob Marshall Wilderness of Northwest Montana. This data, which comes from an area that has managed for an active fire regime for over three decades, allows me to quantify the forest structures resulting from such a fire regime in a mixed-conifer forest ecosystem, as well as describe the potential developmental pathways to each forest structure class. With this chapter, I present a data-driven model of mixed-conifer forest stand structure development under an active fire regime.

All these studies relate to wilderness fire in some capacity. Chapters 1 and 2 primarily focus on fire management as an ecosystem process, while chapter 3 presents the ecological consequences resulting from this fire management approach. The results of these studies, taken together, point to the importance of wilderness areas as valuable sources of information on ecosystem processes and patterns, such as wildland fire and forest structure.

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CHAPTER 1: A history of wilderness fire management in the Northern Rockies

ABSTRACT

Fire management in the United States has defaulted to suppression of most ignitions since 1935. In some large wilderness areas, however, wildland fire has been managed to play a more natural ecosystem role for over four decades. This is true for the three large wilderness areas of the U.S. Northern Rocky Mountains: the Selway-Bitterroot Wilderness, the Bob Marshall Wilderness Complex, and the Frank Church River of No Return Wilderness. Because fire in these wilderness areas is managed so differently from other public lands, this report recounts the most historically important fires managed in these wilderness areas, as well as the lessons learned in terms of managing fire for resource benefit from these fires. This review therefore provides a detailed account of the development of wilderness fire management in the Northern Rockies, including the challenges overcome and the lessons learned from the pioneering days in the 1970s up to present day. An improved understanding of this history can potentially be used to help improve fire management nationally.

INTRODUCTION

Fires in the U.S. Northern Rocky Mountains (hereafter, the Northern Rockies) have been crucial to shaping national fire management policy since the early 1900s. The 1910 fires, which burned over three million acres in the Northern Rockies and threatened multiple towns, were unprecedented in both size and severity for the Euroamerican settlers of the region. Following the devastation of 1910, forest fire management became synonymous with fire suppression. For the newly formed Forest Service, priority was placed on firefighting, with very little thought given to fire management beyond suppression (Pyne 1982). With the New Deal of the 1930s came a build-up of firefighting resources and crews, which set the stage for another national policy shift following the 1934 Selway Fires in Idaho (Koch 1935). Those large fires provided the impetus for adoption of the 10 A.M. policy, which called for full suppression of all wildfires by 10 A.M. the next day (Loveridge 1944, Pyne 1982).

Despite this long history of large fire events in the Northern Rockies catalyzing shifts in national policy towards fire suppression, the region is also home to the White Cap study area in the Selway-Bitterroot Wilderness, which was the first Forest Service wilderness area to allow naturally ignited fires to burn in the early 1970s (DeBruin 1974; Smith 2014; van Wagtendonk

2007). This process of managing, rather than simply suppressing, naturally ignited fires subsequently spread to several wilderness areas throughout the United States, catalyzing a policy evolution to increased fire management for resource benefit (USDA and USDI 2009; van Wagtendonk 2007).

Since the White Cap Study, multiple large wilderness areas in the Northern Rockies have adopted wilderness fire management, or management of lightning-ignited wildfires for ecological benefit. This shift in fire management strategy in Forest Service lands such as the Selway-Bitterroot Wilderness (SBW), the Bob Marshall Wilderness Complex (BMWC), and the Frank Church-River of No Return Wilderness (FCRNRW), as well as Yellowstone and Glacier National Parks, has allowed fire to play a more natural role across vast swaths of land in the Northern Rockies. From these areas, there is a wealth of knowledge to be gained about fire management strategies, fire management effects, and fire ecology.

The purpose of this paper is to compile and synthesize the history, evolution, and key lessons learned from wilderness fire management in the three largest Forest Service wilderness areas in the Northern Rockies. From this history, there is much to be gained in terms of lessons on how to most effectively manage wildland fire for resource benefit. I provide a brief summary of the history of fire management leading up to and following the decision to allow fire in wilderness areas and discuss the setbacks and accomplishments that have occurred in the intervening years. As the first comprehensive history of wilderness fire management in this region, this review provides suggestions as to how to incorporate the lessons learned from these wilderness areas into future fire management policies and decisions.

METHODS

Each of the wilderness areas highlighted in this paper (SBW, BMWC, and FCRNRW) includes an extensive amount of land (1.3, 1.5, and 2.4 million acres, respectively), and all have allowed for some level of wilderness fire management for at least 40 years. They therefore serve as large natural laboratories to study fire management and fire effects on ecosystem.

To synthesize all available information on these wilderness areas, I began with an extensive literature review of peer-reviewed studies, management documents, and Forest Service reports and plans. This provided critical information on both the fire ecology and fire management practices within the Northern Rockies. I also read existing interviews and conducted new interviews with key wilderness fire managers, both retired and current. These interviews were particularly useful in identifying the decision-making processes that contribute to the decisions to suppress or manage a fire in the wilderness, as well as the challenges in implementing this management strategy and the important lessons learned.

To quantify the effects of shifting fire-management strategies, as well as how these shifts interacted with climatic changes, I examined the spatial and temporal differences in area burned across the three wilderness areas. Specifically, I compiled pre-existing fire atlases for the Northern Rockies (Gibson et al. 2014; Parks et al. 2015a,b,c) and updated these atlases to include fires through 2017 using the Region 1 and Region 4 fire perimeter data. I then calculated area burned and fire rotation periods for four time periods representing a different approach to fire management for each wilderness area (Morgan et al. 2008).

Study Area

The Northern Rockies Ecosystem

For the purpose of this review, the Northern Rockies includes northern and central Idaho and western Montana (fig. 1). This area falls within the temperate conifer forest ecosystem of the

Inland Pacific Northwest (Lassoie et al. 1985). The climate is maritime-continental, with short, dry summers and long, cold winters with high precipitation, primarily in the form of snow (NOAA 1985). The diverse topography of the region includes broad valley bottoms and high mountain peaks, which results in a variety of microclimates.

Mountain ranges running largely north to south divide this region. As the prevailing winds come from the west off of the Pacific Ocean, these ranges force the maritime air to rise and release precipitation. Therefore, the western slopes tend to be moister than the eastern slopes of the study region, which are drier and characterized by a continental climate (Arno 1980; Gibson 2006).

Lightning-ignited fires were historically the primary source of disturbance in this region, but the fire regime varies dramatically depending on the climate and topography, which in turn drive the fuel and vegetation types (Agee 1993; Arno et al. 2000). Low-severity, high-frequency nonlethal regimes are characteristic of low-elevation ponderosa pine (*Pinus ponderosa)* and Douglas-fir (*Pseudotsuga menziesii*) forests. Mixed-severity regimes, which dominate in mid-elevation forests and produce patches of live and dead trees, are emblematic of the Northern Rockies, covering approximately 50% of National Forest lands (Arno et al. 2000; Barrett 2004; Quigley 1996). In high elevation and subalpine forests, fire regimes are typically stand-replacing, as the dominant species have very low fire resistance. In this forest type, fire-free intervals average a century or more due to climate conditions that typically result in high fuel moisture (Barrett 2004). Therefore, when high-elevation sites do burn, it generally results in 80-100% tree mortality.

The diversity of fire regimes across the region means that the Northern Rockies were once defined by a mosaic of even-aged and uneven-aged stands at a variety of scales (Ayres

1900; Baker 2009; Leiberg 1900). Vegetation species and structural diversity was high both within and between stands, and this diversity was maintained by the variable fire regimes (Arno et al. 2000; Baker 1993; Keane et al. 1998; Romme 1982; Hessburg et al 2019). Exclusion of fire from the ecosystem by Euroamerican settlers resulted in broad homogenization of Northern Rocky forests, both in stand age and in species composition (Arno et al 2000; Baker 1993; Keane et al. 2002).

Native Americans in the Northern Rocky Mountains

Prior to European settlement, the human population of the Northern Rockies numbered at approximately 30,000 people (Baker 2002). The Flathead, Pend d'Oreille, Kalispel, Nez Perce, and Northern Shoshone tribes inhabited the Northern Rockies west of the Continental Divide, whereas the Blackfoot Confederacy lived in the area east of the Continental Divide (Baker 2002; Waldman 2009). From written records and oral history interviews, these tribes are known to have regularly set fire to the landscape (Barrett and Arno 1999; Stewart 2002). They would ignite fires to influence game drives, promote forage and grazing material, maintain campsites and trails, and communicate amongst tribe members. Such fires were most commonly set in valley bottoms dominated by grassland ecosystems and in low-elevation ponderosa pine or ponderosa pine-Douglas-fir forests (Barrett and Arno 1999).

Because human-ignited fires were largely confined to areas with heavy use, Native Americans likely had a profound but highly localized impact on the fire ecology of the Northern Rockies. This is supported by studies that have found historical fire intervals in valley bottoms much shorter than could be explained by lightning ignitions alone (Arno 1976; Arno 1980; Arno et al. 1997; Baker 2002). Furthermore, a study of climate and human population trends in relation to biomass burned revealed that climate variability largely explained the variability in area burned across Western forests up to the late 19th century (Marlon et al. 2012). Therefore, the sharp decline in Native American burning following increased European settlement in the 1860s likely had the greatest impact in the low elevation, ponderosa pine-Douglas-fir stands of the Northern Rockies (Barrett and Arno 1999).

The Selway-Bitterroot Wilderness

The SBW lies along the border of Montana and Idaho. At 1.3 million acres (790,000 hectares), it is the third largest single wilderness in the contiguous United States (Wilderness Connect). It is administered by the Bitterroot, Clearwater, Lolo, and Nez Perce national forests (fig. 2a). The topography of the SBW is extremely rugged, with over 2,500 m in relief (Rollins et al. 2001).

The climate of the SBW varies from Pacific-maritime in the northwest to continental in the southeast (Finklin 1983). Annual precipitation ranges from 1,000 mm along the Lochsa and Selway Rivers to 1,800 mm in the Bitterroot Mountains. The majority of this precipitation falls as snow, and snowpack at the highest elevations will typically persist into late June (Finklin 1983).

The SBW can be divided into three broad geographic regions. High mountains and forested valleys typify the northwestern portion of the wilderness, which contains the upper portions of the Lochsa and Selway Rivers. Due to the Pacific maritime climate, forest stands here consist of western redcedar (*Thuja plicata*), western hemlock (*Tsuga heterophylla*), grand fir (*Abies grandis*), and western white pine (*Pinus monticola*). High ridges and steep canyons define the central and southern portions of the SBW, whereas the eastern portion is delineated by the Bitterroot Mountains. These steep (3,000+ m of relief) granitic mountains are dissected with east-west running glacial valleys (Rollins et al. 2001; Rollins et al. 2002).

The continental climate of the central, southern, and eastern portions of the SBW results in drier vegetation communities that vary with elevation. Ponderosa pine and Douglas-fir stands characterize the lower elevations forests, which change to Douglas-fir, western larch (*Larix occidentalis*), grand fir, and Engelmann spruce (*Picea engelmannii*) forests within mid-elevation and subalpine forests. Whitebark pine (*P. albicaulis*) and subalpine larch (*L. lyalli*) forests dominate the highest subalpine stands. About 70% of the SBW is subalpine forest (Rollins et al. 2001; Rollins et al. 2002).

Lower elevations of the SBW burn across a range of severities, depending on fire weather and the dominant tree species, and upper elevations burning less frequently, often with high levels of mortality (Brown et al. 1994; Rollins et al. 2002). The fire season extends from June through September, with large thunderstorms common, reaching a peak in July and August (Finklin 1983; Rollins et al. 2000).

Bob Marshall Wilderness Complex

Located entirely in northwest Montana, the Bob Marshall Wilderness Complex comprises three adjacent wilderness areas: the Bob Marshall Wilderness, the Great Bear Wilderness to the north, and the Scapegoat Wilderness to the south. Together these areas add up to over 1.5 million acres (610,000 hectares). Combined with the additional 1 million acres of roadless land surrounding the complex, the BMWC constitutes one of the most remote areas in the contiguous United States. The Great Bear Wilderness is managed completely by the Flathead National Forest; the Bob Marshall by the Flathead and the Lewis and Clark National Forests; and the Scapegoat by the Helena, Lewis and Clark, and Lolo National Forests (fig. 2b). The complex as a whole is characterized by mountainous terrain dissected by generally north-south oriented river drainages (Keane at al. 1994).

The BMWC straddles the Continental Divide, which plays a large role in shaping the climate of the complex. West of the divide, the climate is modified maritime, with cool wet winters and warm dry summers. Precipitation here can vary from 500 mm/year in some valleys to 2,750 mm/year on the Swan Front to the very west of the Bob Marshall Wilderness (Keane et al. 1994). To the east of the divide, the climate is continental, with generally colder winters and warm dry summers. However, winter temperatures vary widely, and high winds persist throughout the year. On the east side, precipitation ranges from 400 mm/year to 1,500 mm/year (Keane et al. 1994).

Douglas-fir and lodgepole pine tend to dominate forest stands in the low elevations of the BMWC. Western larch is also common in these low-elevation forests west of the Continental Divide, whereas aspen (*Populus tremuloides*) becomes more much prevalent east of the divide. In very moist, low-elevation areas in the far northwest of the complex, western hemlock and western redcedar are also common. At higher elevations, lodgepole pine, subalpine fir, and Engelmann spruce become more prevalent. The highest elevation stands contain whitebark pine and subalpine larch at treeline (Cansler et al. 2018).

The majority of the BMWC is cold, moist forests with only small patches of ponderosa pine stands in the drier valleys and river terraces (Östlund et al. 2005). The predominant fire regime is one of infrequent, mixed- to high-severity burns (Keane et al. 2006; Teske et al 2012). The fire season typically runs from mid-July through September (Parks et al. 2015d). Towards the end of the season, the polar vortex will frequently migrate south, bringing strong winds that can cause quick increases in the size and severity of fires (Teske et al. 2012).

Frank Church-River of No Return Wilderness

The FCRNRW is located entirely within Idaho, just to the south of the SBW. The two wilderness areas are separated only by a narrow road, the Magruder Corridor. At 2.4 million acres (1,000,000 ha.), the FCRNRW is the largest contiguous wilderness area in the lower 48. Management of the FCRNRW is split between four national forests: The Salmon-Challis, Payette, Bitterroot, and Nez Perce-Clearwater National Forests (fig. 2c). As the most southerly of the Northern Rockies wilderness areas covered in this review, it is typically drier than either the SBW or the BMWC (Teske et al. 2012).

The FCRNRW is defined by extreme environmental gradients. River breaks and canyons dissect the wilderness, with dry ponderosa pine grasslands and sagebrush dominating the lower elevations at the canyon bottoms and lodgepole pine, subalpine fir and Engelmann spruce characterizing the upper elevations (Teske et al. 2012). Overall, ponderosa pine-Douglas-fir forests make up the majority of the wilderness area (Barrett & Arno 1988). Precipitation in this rugged wilderness ranges from 380-430 mm at the lowest elevations to 1,300-1,500 mm at the highest elevations in the western mountains (Finklin 1988). Most of this precipitation falls in the form of snow (Finklin 1988).

A mixed-severity fire regime characterizes the majority of the FCRNRW. However, due to the drier climatic conditions there is a higher preponderance of the low-severity, high-frequency fire regime as compared to the SBW and BMWC (Arno 1980). The fire season tends to run from early July to mid-September (Parks et al. 2015d), with the lightning season running from May to September and peaking in June through August (Teske et al. 2012). Lightning-caused fires are especially common in the river breaks where dry conditions and fine fuels are prevalent (Arno 1980). Fire management within the FCRNRW is complicated by the presence of multiple large inholdings, which are primarily located along the major river corridors (Irey 2014).

Wilderness Fire Management Timeline

I divided the history of fire management for each wilderness area into four eras: Pre-Exclusion, Exclusion, Transition, and Fire Management. These eras are delineated by changes in fire management policies as well as shifts in climate (fig. 3, Higuera et al. 2015; Morgan et al. 2008). During the Pre-Exclusion period, which is defined as the period up to and including 1934, fire suppression activities did occur but were largely ineffective due to the lack of manpower and technology (fig. 3; Koch 1935; Morgan et al. 2008; Pyne 1982). This was especially true during the climate-enabled and weather-driven regional fire years in the Northern Rockies, including 1910, 1919, 1926, 1929, 1931, and 1934 (Morgan et al. 2008).

The Exclusion Era begins in 1935, with the installation of the 10 A.M. policy, and ends between 1970-1979, depending on the wilderness (fig. 3). During this time, funding and equipment for firefighting was plentiful and national policy demanded aggressive fire suppression of all new ignitions (Loveridge 1944; Pyne 1982). As a result, no regional fire years occur during this period, and in fact the maximum area burned in the Northern Rockies during this period was 68,000 hectares (Morgan 2008). This is likely due to an interacting effect of fire management and climate, as this period also saw cooler springs and few very dry summers (Morgan et al. 2008).

The Transition Era represents a period when thinking on fire management began to shift. During this time, more wilderness and fire managers recognized the ecological importance of fire to the forests of the West (Habeck and Mutch 1973, Wright and Heinselman 1973a). The start dates for this period vary among the wilderness areas, depending on when each enacted a plan for wilderness fire management (fig. 3). In each case, this process depended on a few key land managers recognizing the ecological necessity of fire and then conducting the necessary studies and planning processes to allow for fire. This period is delineated from the fire management period, however, because only a small portion of each wilderness area was allowed for fire management during the transition period. As a result, very little acreage actually burned. For this reason, I include the transition era with the exclusion era when calculating fire rotation periods.

The introduction of wilderness fire to the entirety of the SBW in 1978 ushered in the Fire Management era. The fire management era for each individual wilderness begins at a different time, depending on when a wilderness fire plan was enacted for the entirety of that wilderness area. Overall, however, this era is marked by increasing flexibility in fire management policy allowing for increased fire management for resource benefit. Early on in the fire management period, the climate remained cool and moist, which kept fire activity low and allowed wilderness fire programs to develop with relatively few large fire events. As a result, there are no regional fire years from 1978-1987, a period also marked by a dramatic increase in the amount of land allowing for wilderness fire in the Northern Rockies (Morgan et. al 2008; van Wagtendonk 2007).

Since 2000, a hotter, drier climate has increased the length of fire seasons in the Northern Rockies, resulting in larger area burned (Abatzoglou and Williams 2016; Higuera et al. 2015; Holden et al. 2018; Jolly et al. 2015; McKenzie and Littell 2017; Westerling 2016). At the same time, fire management policy has continued to evolve to allow for more management of fire for resource benefit. Most recently, the 2009 policy review removed the distinction between a suppression fire event and a fire that is managed for resource benefits. Instead, more than one objective is permissible for any individual fire, which has allowed for more flexibility in fire management both inside and outside wilderness areas (USDA and USDI 2009).

Prior to 2009, however, fires managed for resource benefit were first called Prescribed Natural Fires (PNFs) and then Wildland Fire Use events (WFUs, Hunter et al. 2014; van Wagtendonk 2007). Since 2009, any wildland fire managed with non-suppression strategies is frequently referred to as a fire managed for resource benefit (Harbour 2010). With the multiple changes in policies and terminology, it can be confusing to talk about fires that burn in wilderness areas. Therefore, for the purposes of this review, all fires managed within a wilderness using a strategy other than full suppression will be referred to as "wilderness fire."

NATIONAL HISTORICAL CONTEXT (PRE-1972)

The expansion of Euroamericans across of the western United States during the 19th and 20th centuries drastically altered the fire regime of North America. With settlers came an initial increase in burning, due in part to fires set for land clearing as well as the increased spark production that accompanied expanding railroad use (Marlon et al. 2012; van Wagtendonk 2007). In addition, increased forest harvesting during this time altered fuel loads, further raising fire activity (van Wagtendonk 2007). Fires were only fought when they threatened settlements, so wildland fire management did not become a national issue until the formation of the National Park System in 1872. At this point, the U.S. Army assumed responsibility for fire suppression in the first national parks, such as Yellowstone, Yosemite, and Sequoia, and fire suppression remained the default option with the formation of the National Park Service in 1916 (van Wagtendonk 2007). With the establishment of the Forest Service in 1905, fire suppression became an even higher priority, as the nascent organization viewed fires as a waste of valuable timber (Pyne 1982; van Wagtendonk 2007).

The Forest Service faced early threats from anti-conservationists in Congress (Egan 2009; Steen 1976). The record-setting fires of 1910, which were both expansive and deadly, therefore gave the young organization a renewed purpose and made fire suppression a primary objective of forest management nationwide (Arno and Allison-Bunnell 2002; Pyne 1982; Pyne 2016). The concept that all fires are bad, however, was never universally embraced. As early as 1916, the Assistant District Forester for California, Roy Headley, instituted a program that allowed for low intensity fires in remote areas. He argued that, unless fire threatened high-value timber, the cost of suppressing these remote fires was too high to justify the economic values saved (Pyne 1982; van Wagtendonk 2007).

This debate over fire suppression in backcountry, or "low value" lands, continued throughout the first half of the 20th century. During this time, the Forest Service and other land management agencies were pursuing fire suppression, but the efficacy of this approach was limited in the backcountry due to the high cost of moving firefighters and equipment into the more remote areas. The Forest Service therefore organized the Low Value Land Expedition in 1932. For this trip, several Washington office representatives and regional foresters rode through central Idaho wilderness areas on an extended pack trip. Accompanied by the forest supervisors and district rangers who managed these remote areas, the group debated over whether to continue with full suppression in the backcountry, or to adopt a more moderate approach of allowing some fires to burn where natural barriers would contain fire spread (Larson 2016; USDA 1932).

Around the same time, a similar debate was happening in the professional forestry literature. On one side was Elers Koch, who by 1935 was serving as the Assistant Regional Forester for Region 1 (Larson 2016). Koch had worked in the Northern Rockies for nearly his whole career, and fought in the fires of 1910, 1919, 1929, and 1934. After seeing firsthand the futility of attempting to suppress the big backcountry blowups, Koch opposed the buildup of roads, airplane landing fields, and telephone lines in backcountry areas in his essay “The passing of the Lolo Trail” (Koch 1935). In this essay, which was published in the Journal of Forestry with support from wilderness advocates such as Bob Marshall and Aldo Leopold (Pyne 1982), Koch argued that low value areas would be best managed without further development and fire control. Instead, they should be left “pretty much to the forces of nature” (Koch 1935).

Countering Koch and the wilderness advocates was Earl Loveridge, the Assistant Chief of the Forest Service. In rebuttal to Koch’s article, Loveridge published “The opposite point of view” in the same issue of the Journal of Forestry (Loveridge 1935). In this essay, he argued for the other extreme as a fire management policy: that all fires should be put out by 10 A.M. the day following detection. This, he argued, was entirely possible with the correct techniques, policies, and resources (Loveridge 1935; Pyne 1982). Eventually, Loveridge’s argument won out. The buildup of firefighting crews and equipment that accompanied the New Deal, combined with the recent memory of the 242,000 acres that burned in the Selway Fires of 1934, led to the passage of the 10 A.M. policy in 1935 (Silcox 1935; Pyne 1982).

The National Park Service was the first agency to actively push back against this blanket policy of fire suppression on all federally managed lands (van Wagtendonk 2007). In 1950, the Kaweah Basin in Sequoia National Park was designated as a research area where some fires would be allowed to burn (Rothman 2007). About a decade later, in Yosemite National Park, the Assistant Chief Ranger wrote a recommendation to allow fires to burn at elevations greater than 8,000 ft (2,438 m) because fuels at these elevations were rarely heavy enough to produce large fire events (van Wagtendonk 2007). Although neither of these management shifts resulted in many acres burned, they did reveal a shift in thinking. For both, the rationale expanded beyond the economics of firefighting to include the ecological benefits of fire (van Wagtendonk 2007).

This change in thinking was echoed in the 1963 Leopold Report, in which an advisory committee recommended that National Parks take an ecosystem approach to park management (Leopold et al. 1963). This opened the door for fire management beyond suppression, and by 1972 Sequoia, Kings Canyon, Saguaro, and Yosemite National Parks had all adopted some form of natural wildland fire management (Hunter et al. 2014; van Wagtendonk 2007).

Around the same time, researchers began exploring the historical role of fire in wilderness areas and emphasizing the importance of fire to vegetation communities (Gabriel 1976; Mutch 1970; Romme 1982). In fact, a special issue of Quaternary Research in 1973, based on a series of papers presented in an organized symposium held during the joint Ecological Society of America and American Institute of Biological Sciences annual meetings in August, 1972, was dedicated to the role of fire as an ecosystem process and the future of fire management in wilderness ecosystems (Wright and Heinselman 1973a). The introduction to this special issue, in an attempt to highlight the shortcomings of Clementsian ecological theory in fire-dependent systems, outlines the critical role of fire in reducing fuel accumulation, shaping vegetation communities, and regulating ecosystem processes such as nutrient cycling, culminating in an appeal to integrate prescribed and naturally ignited fire in wilderness areas (Wright and Heinselman 1973b).

This larger shift in how fire and fire management was viewed, combined with the passage of the 1964 Wilderness Act, paved the way for the Forest Service to begin reconsidering its own universal acceptance of the 10 A.M. policy. In 1969, a wilderness workshop assembled in Missoula, Montana to discuss the possibility of allowing fire in wilderness areas (van

Wagtendonk 2007). The White Cap Study in the Selway-Bitterroot Wilderness, which began contemporaneous with this movement in the scientific community, paved the way for fire management for resource benefit in Forest Service wilderness areas across the United States.

WILDERNESS FIRE MANAGEMENT IN THE NORTHERN ROCKIES

Selway-Bitterroot Wilderness

Pre-Exclusion (up through 1934)

A variety of data sources have been used to reconstruct the fire regime of the Selway-Bitterroot Wilderness prior to aggressive fire exclusion. In 1898, the U.S. Geological Survey (USGS) contracted John B. Leiberg to survey the Bitterroot Forest Preserve. His report details the geology, vegetation, and fire history of the area, which encompasses the modern-day SBW. A map of fire-affected areas within the preserve, which is much larger than the current wilderness, suggests that very little of the area was unburned from 1719 to 1898 (Leiberg 1900). From Leiberg's maps, Habeck (1977) determined that approximately 35% of the Selway River drainage burned from the 1860s to 1898.

Fire scar studies in this area also reveal the dominant role of fire in shaping this landscape. Within the Bitterroot National Forest, tree scars revealed a frequent fire return interval for both ponderosa pine and Douglas-fir habitats (averaging once every 6-20 years), while vegetation types found on higher elevation, moister sites burned less frequently (averaging once every 20-40 years, Arno 1976).

The GIS data for the SBW also indicate extensive fire during the pre-exclusion era (fig 4). The polygon fire history compiled from the national forests indicates that 443,213 hectares (1,095,203 acres) burned from 1889-1934 (Gibson 2014; Morgan et. al 2017). This polygon history suggests that the fire rotation period (FRP) for the entire SBW, or the time necessary to burn an area equal in size to the study area, during the pre-exclusion period was 55 years (table 1).

Native American populations in the Northern Rockies region were likely largest during the pre-exclusion era, especially prior to increased Euroamerican settlement in the 1860s (Barrett and Arno 1999). Native American land use within the SBW can be identified from studies of bark peeled trees. These trees bear a distinctive scar from the Native American's harvesting of the inner bark, which provided an important source of fiber and vitamin C in a diet otherwise heavy in animal protein (Östlund et al. 2005; Östlund et al. 2009). In the Bitterroot Mountains, sampling of trees containing these bark peel scars revealed Native American resources use in the area as recently as the 1930s. This suggests that Native Americans shaped the fire ecology of the low-elevation forests into the early 20th century (Josefsson et al. 2012). Barrett's (1977) research into the primary Native American living and traveling areas suggests that Native American influence on fire activity was likely greatest along the east slope of the Bitterroot Mountains.

Fire Exclusion (1935-1969)

The SBW saw dramatically less fire activity following the implementation of the 10 A.M. policy, coupled with the cooler, wetter weather and climate typified by the fire exclusion era (fig. 3). During this period, only 10,439 hectares (25,795 acres) burned over the total wilderness area, resulting in a dramatically increased FRP of 2,194 years (table 1). Forest Service fire records indicate that fire starts were still common, but most were suppressed before exceeding an acre (Habeck 1977). This resulted in greater vegetation homogeneity, as denser, older forest stands replaced various early successional stages (Habeck 1977).

Transition (1970-1977)

The concept of wilderness fire management in National Forests was born in the SBW. Specifically, wilderness fire management began with some radical thinking by a few Forest Service employees who were questioning the wisdom, practicality, and ethicality of suppressing fire in wilderness areas, and saw the SBW as a good testing ground for a new approach to wilderness fire management.

The spark of the idea came from William "Bud" Moore, the Region 1 Director of Fire Control from 1967-1974. Moore had grown up and lived nearly his whole life in western Montana (Moore 1996). Despite having no formal training in forest or fire ecology, he recognized from his hunting and trapping trips that the exclusion of fire from the Northern Rocky Mountains had fundamentally changed the forest and fuels structure of the region (Smith 2014). He also recognized that the Wilderness Act of 1964 made fire suppression in the wilderness "practically illegal" (USDA 2002b), as it called for wilderness areas to be "affected primarily by the forces of nature, with the imprint of man's work substantially unnoticeable" (Wilderness Act 1964). He therefore began to push for a more ecologically informed view of wilderness fire management in the Northern Rockies.

To achieve this goal, Moore gathered a team that, including himself, became known as the "White Cap Five." This team included Bill Worf, the Chief of Recreation and Lands for Region 1 who had argued for an end to complete fire exclusion under the premise that it was ecologically unnatural (Smith 2014). It also included Orville Daniels, who at the time was the supervisor for the Bitterroot National Forest. Daniels had an interest in fire and was willing to take the risk of allowing fire within his jurisdiction (USDA 2002b). Rounding out the team were Dave Aldrich and Bob Mutch, whom Daniels hired to co-lead the project. Aldrich had worked in fire control in Idaho, and Mutch had been a smokejumper and then a research scientist at the Fire

Sciences Laboratory (Fire Lab) in Missoula. While at the Fire Lab, Mutch conducted research which concluded that some plant ecosystems actually depended on fire for survival and reproduction, and therefore burned more readily than other plant communities (Mutch 1970). This team was fully assembled by the summer of 1970 (Smith 2014).

Bud Moore and Orville Daniels settled on the White Cap and Bad Luck drainages of the SBW (hereafter, the White Cap study area), as the location for their experiment. Those drainages were chosen because they represented a microcosm of the wilderness as a whole in terms of topographical features and vegetation communities. They also were contained by a "granite firewall" to the east, or high rocky country that made containing fires within the drainage easier (USDA 2017).

Beginning August of 1970, Aldrich and Mutch conducted extensive surveys of the vegetation and fuels within the White Cap study area. They established 380 plots for these surveys, many of which were re-measured each year over the three-year planning period. In addition to the vegetation and fuels surveys, Aldrich and Mutch listed every plant species within the study area, conducted bird species inventories, and collected and described the hydrological and geologic features of the drainage (Smith 2014).

Following three years of data collection and planning, Aldrich and Mutch devised a fire management plan for the White Cap study area. They divided the area into five ecological land units (ELUs), which were defined based on similarities in topography, vegetation, fuels, and fire potential (USDA 2002b; Mutch 1974). Each ELU received its own fire management prescription, which provided rules as to when to allow fire in each of the zones (table 2, Mutch 1974). Beyond providing a management plan for the White Cap study area, these guidelines were intended to provide a model for fire prescription development in wilderness areas across the country (Aldrich and Mutch 1973, Moore 1974, Smith 2014).

Immediately following the approval of the White Cap fire management plan in the Washington Office in August of 1972, a lightning strike ignited a fire in the shrubfield unit of the Bad Luck Drainage. Because it was within the “observation” prescription for that area, the fire was allowed to burn with no suppression or direct management actions. Due to the wet conditions of that season, the Bad Luck Fire burned itself out at a size of approximately 24 by 24 feet (fig. 5, Mutch 1974).

The summer of 1973, however, was much hotter and drier than 1972. Lightning struck this time on August 10th in the ponderosa pine-savanna zone within the Fitz Creek Drainage. The prescription for this ignition, named the Fitz Creek Fire, was suppression along the east flank of the fire where it threatened to cross the study area boundary, while the rest of the fire was observed (Daniels 1974; Mutch 1974).

On August 15th, wind conditions caused the Fitz Creek fire to spot across White Cap Creek, which served as the boundary for the White Cap study area. This new fire was outside the study area, and therefore was not included under the fire management plan. As a result, it was given its own name, the Snake Creek Fire, and managed with full suppression tactics (Daniels 1974). Ultimately, Snake Creek was suppressed at 1,600 acres on August 21st, whereas fall rains extinguished the Fitz Creek fire on September 21st at approximately 1,200 acres (Daniels 1974). The footprint for Fitz Creek burned across the ponderosa pine savanna, shrubfield, and ponderosa pine-Douglas-fir ELUs (Mutch 1974). Five other fires were ignited by lightning that summer and were managed with a combination of observation, observation and suppression, and suppression. All remained under a quarter of an acre in size (Mutch 1974).

The fires of 1973 were the first big test of the White Cap fire management plan. That summer provided some important data on wildfire impacts on vegetation structure, fuel loading, and wildlife habitat, as well as lessons for wilderness fire managers. Ecologically, the Fitz Creek fire created heterogeneity, with unburned islands interspersed amongst patches of low to high severity areas (Daniels 1974). Within the burnt areas, fuel levels were largely reduced immediately following the fire, with the exception of young Douglas-fir stands, where scorched needles more than doubled surface litter loads (Mutch 1974). Shrubs quickly re-sprouted, and both mammals and birds were observed within the burn perimeter both during and immediately post-fire (Mutch 1974).

Field crews sampled fuel levels and vegetation data from 1973 to 1977, and again in 1980, on permanent plots within the burn perimeter (Smith 2014). Of those initial 380 plots sampled to produce the fire prescriptions, 100 were re-sampled following the reintroduction of fire. These surveys allowed for a better understanding of the long-term effects of wildland fire in fire-dependent ecosystems, both by fire managers and the public at large (Smith 2014).

The 1973 fire season also provided the first example of a successful Forest Service wilderness fire management approach. The Fitz Creek Fire and the Snake Creek Fire increased land manager's confidence in the prescriptions assigned to the ELUs, as these two fires had burned through four of the five vegetation types in the White Cap study area (Daniels 1974). They also learned that when making fire management decisions, fire behavior predictions allowed for a better assessment of risk than the number of acres burned (Daniels 1974).

After the 1973 fire season, the Whitecap managers were able to make recommendations to other wilderness areas attempting to institute a fire management plan. They recommended: 1) Integrating the fire management plan into the overall land use plan; 2) Promoting inter-agency cooperation when managing wildfires; 3) Informing the public on the natural role of fire; 4) Educating fire specialists and land managers on the ecology of fire; and 5) Working current knowledge on fire ecology into institutional guidelines and policies (Moore 1974).

Fire Management (1978-present)

In 1976, relying on the lessons learned from the 1973 fire season, the Bitterroot, Clearwater, Lolo, and Nez Perce National Forests drafted a fire management plan for the entire SBW (Yurich 1976). This plan was approved in 1978, the same year that the 10 A.M. policy was abandoned at the national level (Pyne 1982). The new Forest Service policy towards wildfire allowed for managing active fires for resource benefit, versus a suppression-only tactic, especially in wilderness and backcountry areas.

The 1979 fire season, and the Independence Fire on the Moose Creek Ranger District in particular, provided the new SBW fire management plan with its first big test (Keown 1985). During the summer of 1979, 59 fires ignited on the Moose Creek Ranger District, 55 of which were lightning caused. The Independence Fire, one of the 10 lightning-ignited fires that were then designated as "prescribed natural fires," eventually growing to burn over 16,300 acres over the course of 106 days (Keown 1985).

The duration and sheer size of the Independence Fire tested the ability of the SBW fire managers to manage a wilderness fire. Within the projected fire perimeter, three trail bridges and a private landholding were threatened and therefore required suppression tactics (Keown 1985). Rothermel fire models, which were used to predict fire intensity and rate of spread, proved to be crucial when planning suppression and burn-outs for point protection (Keown 1985). This fire thereby increased managers' confidence in the fire behavior models. It also achieved the ecological goals outlined in the fire management plan, as vegetation diversity increased following the Independence Fire due to the variable timing and severity of the fire (Keown 1985).

The Independence Fire also taught managers the importance of humility when dealing with wilderness fire events (George Weldon, Retired U.S. Forest Service, Missoula, MT, personal communication, December 5, 2018). Although the fire models helped with planning some limited suppression activity, there was ultimately no way to predict the large size of the fire when it ignited in early July. Fire managers who worked on the Independence Fire therefore walked away with the understanding that fire is unpredictable, and that wilderness fire managers must both accept this risk and then minimize it to the greatest extent possible with the tools available (George Weldon, Retired U.S. Forest Service, Missoula, MT, personal communication, December 5, 2018).

The early years of fire management within the SBW were controversial, but also provided the crucial testing ground for wilderness fire management. Not only did the program survive challenging events such as the escape of the Snake Creek Fire, but the early successes in managing relatively large fires with no loss of human life or property provided crucial evidence indicating that wilderness fire was not as heretical an idea as it had first seemed. With these successes, based off careful data collection and planning, other fire managers now had a roadmap for developing a wilderness fire management program. In addition, the research conducted in the White Cap study area provided managers with data that showed wilderness fires could achieve the ecological objectives of increasing landscape heterogeneity and reducing fuels on the ground (Smith 2014). In short, this experiment within the SBW showed the fire management community and the public that fire did not mean death for forests, but rather renewal.

The 1980s and 1990s saw minimal fire activity in the SBW, with the important exception of 1988. As was the case for most of the Northern Rockies, 1988 was a highly active fire year for the SBW, with over 49,000 acres burned (Brown et al. 1994). The amount of area burned was more than double what the SBW had experienced since the introduction of wilderness fire management in 1972. In addition, the resulting national fire policy review of 1989 impacted fire management at the national level. Following this review, more complexity was added to the decision-making process required for whether or not to manage an ignition as a wilderness fire (Byron Bonney, Retired U.S. Forest Service, Missoula, MT, personal communication, November 27, 2018). This complexity, however, was necessary to continue to build the wilderness fire program, as it required that fire managers completely think through the possible consequences of fire behavior and more completely consider safety and values at risk (Byron Bonney, Retired U.S. Forest Service, Missoula, MT, personal communication, November 27, 2018).

The public reaction to the 1988 fires also resulted in more hesitancy around wilderness fire management within the Regional and National Offices, especially during large fire years (Jack Kirkendall, retired U.S. Forest Service, Missoula, MT, personal communication, December 3, 2018). Therefore, until the mid-1990s, fire managers hoping to allow fires to play their natural role in wilderness areas would often label them as suppression fires but not take any direct management action on the fire, essentially treating it as a wilderness fire event (Byron Bonney, Retired U.S. Forest Service, Missoula, MT, personal communication, November 27, 2018).

The summer of 1994 was another exceptional season during these quiet decades. This was the most active fire season to date for Region 1, and the high activity in the SBW during August of this year came on the heels of the South Canyon Fire in the Grand Junction BLM District of Colorado. The South Canyon Fire started in July of 1994, ultimately killing 14 firefighters and prompting heightened national attention to firefighter safety (Bonney 1998; van Wagtendonk 2007). The high national preparedness levels and drought conditions for the SBW meant that all ignitions in August of 1994 were declared suppression events. However, the increased commitment to firefighter safety, as well as the scarcity of fire-fighting resources at the national level, meant that many of these suppression events on the SBW were managed with alternative suppression strategies that largely let the fires burn within the wilderness boundary (Bonney 1998).

The South Canyon Fire also prompted a major review and update of national wildland fire fighting policy. This review ultimately reaffirmed management of wildfire for resource benefit while catalyzing the development of guidelines that emphasized the importance of planning explicitly for fire. These guidelines included recommendations that all federal land units subject to wildland fire develop specific plans for appropriate fire response, as well as a description of the protocols and plans necessary in order for land management agencies to manage fire for resource benefit (USDA and USDI 1995). Ultimately, having these guidelines in place revitalized wilderness fire management across the nation, including within the SBW, because it gave managers an explicit, sanctioned pathway for decision making (van Wagtendonk 2007).

In 1997, Dave Campbell took over as the District Ranger for the West Fork Ranger District of the Bitterroot National Forest. Campbell was greatly influenced by the vision of the White Cap Five, and Bud Moore in particular. He felt the drive to continue the legacy that Moore had established for wilderness fire management. This drive, combined with the revitalization of wilderness fire following the policy review of 1995, resulted in much more wilderness fire management on the SBW (Bob Mutch, Retired U.S. Forest Service, Missoula, MT, personal communication, November 19, 2018).

Campbell's mentality was that there should be no hard number of fires or acres burned that would serve as a cut-off for the go-no go decision that has to happen after each lightning ignition. Instead, he considered each fire individually as a potential tool to accomplish resource-management goals (Dave Campbell, Retired U.S. Forest Service, Hamilton, MT, personal communication, November 28, 2018). Because of this commitment to fire as a natural part of the ecosystem, over 300 lightning-ignited fires were managed as wilderness fires during Campbell's 17-year tenure (Bob Mutch, Retired U.S. Forest Service, Missoula, MT, personal communication, November 19, 2018).

During Campbell's time there were some large fire years that both challenged and changed wilderness fire management in the Northern Rockies. In 1998, for example, there were 20 fires managed for resource benefit on the West Fork District alone (Dave Campbell, Retired U.S. Forest Service, Hamilton, MT, personal communication, May 21, 2018). According to Campbell, as these fires grew, questions began to emerge about how to deal with fires that crossed the Magruder Corridor, a road that separates the SBW from the FCRNRW and is classified as a non-wilderness area (Dave Campbell, Retired U.S. Forest Service, Hamilton, MT, personal communication, May 21, 2018). Since these fires had technically "escaped" the wilderness, even if just briefly, were they still able to be managed as wilderness fire events?

Additionally, at the time every wilderness fire was given a "maximum manageable area" (MMA), which was defined as a designated, geographic boundary that set the limits of a wilderness fire in terms of the ability to both achieve ecological management goals and minimize fire risk. Management plans required full suppression for a fire that crossed an MMA boundary.

The fires of 1998, however, caused managers such as Campbell to question why an MMA boundary should automatically be placed at a wilderness boundary such as the Magruder Corridor. With these arbitrary boundaries, there was often no ecological justification and crossing the boundary posed little risk to human life or property. These questions helped to shape future wilderness fire management decisions in the region over the next decade.

The fire season of 2000 was another big year for Region 1, and particularly for the populated Bitterroot Valley of western Montana. The 2000 fire season followed a three-year period of dry weather and above-average temperatures. That summer, many fires started and spread quickly across the Northern Rockies following a series of lightning storms and strong winds (Harmon et al. 2001). On the Bitterroot National Forest alone, 307,000 acres burned in what became known as the Valley Complex, with 3,200 acres burning in the SBW (USDA 2000). Outside the forest, 49,000 acres burned, and 240 structures were destroyed (Backus 2005; USDA 2000).

Fire behavior in 2000 differed dramatically between wilderness and non-wilderness areas. While the fires near Sula and Darby in the Bitterroot Valley were largely high severity, plume-dominated events, the fires within the wilderness burned in a mosaic pattern (Stu Hoyt, Retired U.S. Forest Service, Hamilton, MT, personal communication, November 28, 2018). In many cases, the 2000 fires ran into previously burned areas, which either slowed or stopped the rate of fire spread (USDA 2000; USDA 2002b). As a result, not a single fire escaped the wilderness boundary, which was a serious concern for the those fighting fires in the valley (Orville Daniels, personal communication, retired U.S. Forest Service, Missoula, MT, personal communication, November 5, 2018). These anecdotal stories from the Valley Fire Complex indicated that the wilderness fire program was effective at reducing fuel loads and fire spread, a result that has since been supported by spatial analysis of the limiting effects of previous fire on subsequent wildland fire spread (Parks et al. 2014, 2015d; Teske et al. 2012).

The largest fire year in the SBW during Campbell's time as District Ranger, however, was 2005. Once again, Campbell placed a heavy emphasis on allowing fires to burn, and by the end of the fire season he had approved 50 fires as wilderness fires (USDA 2017). The neighboring FCRNRW similarly had an active fire season with at least 20 fires being managed for wildland fire use (Parks 2006). With all these fires, the questions about the efficacy of the MMA designation raised during and after the 1998 season came to a head. A wildland fire use working group was called together to address management decisions in the SBW, FCRNRW, and the portions of surrounding national forests that allowed for wildland fire management for resource benefit. The result of the working group was a "mega" MMA, or a boundary drawn around the approximately 4 million acres of land between and surrounding the two wilderness areas that allowed for wildland fire management for resource benefit. This ensured that a fire would not be reclassified from a wilderness fire to a wildfire, and therefore suppressed, just because it crossed an administrative boundary (Parks 2006).

The mega-MMA decision, however, also raised some concerns about how managers would make a no-go decision on a fire within such a large and potentially burnable area. They felt the situation could easily get out of hand (Parks 2006). Proponents of the mega-MMA emphasized that coordination and communication among the various administrations, combined with careful consideration of each fire event individually, would ensure the success of this management approach (Parks 2006). Indeed, the mega-MMA proved useful in 2005 for the SBW and FCRNRW and ultimately was a good testing ground for managing wilderness fires that crossed boundaries and therefore required effective collaboration and communication (Parks 2006).

These lessons learned from the mega-MMA classification were reflected in the 2009 Guidance for Implementation of Federal Wildland Fire Management Policy (USDA & USDI 2009). Within this policy guideline, the distinction between a wildland fire use event and a wildfire was eliminated, leaving only two types of classification for fires: prescribed fire and wildfire (USDA & USDI 2009). Additionally, any wildfire could be managed for multiple objectives, depending on the characteristics of the fire and the values at risk (USDA & USDI 2009). As a result, fire management became more flexible, both inside and outside wilderness areas.

In the current era, fire management in the SBW requires acceptance of the reality that not all fires will be small, low-intensity fires. This became evident on a national level with the 1988 fires in Yellowstone National Park and continues to hold true as area burned within the SBW remains high at least once every several years (Dave Campbell, retired U.S. Forest Service, Hamilton, MT, personal communication, May 21, 2018, fig. 3). As longer duration and larger wilderness fire events become more common, the SBW continues to serve as a leading example for fire management for resource benefit.

The SBW's long history of managing fire for resource benefit made wilderness fire management easier within its boundaries during the early 2000s, as previously burned areas often slowed or stopped the spread of active fires (Parks et al. 2014; Teske et al. 2012; USDA 2017). Furthermore, the extensive experience with wilderness fire has created a culture in which the natural process of fire is valued, both as a land management tool and as an end unto itself, and suppression is viewed as a missed opportunity (Beckman 2008). The rich history of wilderness fire in the SBW has made it an important laboratory, both for studying the effects of wilderness fire in a Northern Rockies ecosystem as well as the best practices for managing fire within a wilderness area.

Current conditions

Since the White Cap Study, fire has played a prominent role on the landscape of the SBW (fig. 4 & fig. 6). Since 1978, almost half of the wilderness area burned at least once, with approximately 11% burning multiple times over this period (table 3). As a result, the vegetation composition, structure, and heterogeneity are markedly different compared to areas where fire has been consistently excluded.

At the landscape scale, vegetation mosaics have replaced the even-aged, homogenous stands that tend to dominate in the absence of fire (fig. 7). This is readily apparent within the White Cap study area, which has seen multiple fire events since 1972 (fig. 7a). Across the wilderness, however, this prevalence of vegetation mosaics is evident wherever wilderness fires have been allowed to burn over the past several decades (fig. 7b).

At the stand scale, the impacts of fire are also clearly evident, as it has created open, park-like stand structure in many ponderosa pine stands at low elevations (fig 8a). Western larch forests, which have historically experienced mixed-severity fire regime, have similarly seen a clearing out of the understory in many locations, while the large, fire-resistant trees tend to remain (fig 8b).

Bob Marshall Wilderness Complex

Pre-Exclusion (up through 1934)

The BMWC was surveyed as part of the Lewis and Clark Forest Preserve late in the 19th century for the USGS. H. B. Ayres, who conducted the survey in 1899, visually estimated that

914,000 acres of the 2,965,000-acre preserve had burned within the past 40 years; a number he acknowledged likely underestimated the total area affected by past fires. Furthermore, the maps published with the report show that what is now the BMWC had far fewer of the continuous forest tracts that typify the wilderness today (Ayres 1900). A report on forest fire in the Bob Marshall Wilderness written in the 1960s further emphasizes the importance of fire to the ecology of the BMWC. In the report, Steele (1960) highlights the effects of the large fires of 1889 and 1910, which he estimates burned 35% of the land in the BMW.

A tree ring analysis using both fire scars and stand age, conducted by Gabriel (1976) to determine the fire history of the Danaher Creek drainage in the southeastern portion of the Bob Marshall Wilderness, further supports the active role of fire in the BMWC described by Ayres. The analysis estimated that the natural fire rotation period for the Danaher Creek drainage, from c. 1749-1946, was about 150-200 years (Gabriel 1976). These fires were of mixed-severity and extent, with some years characterized by large, severe fires and others by less severe, surface fires. A large fire occurred somewhere in the drainage each decade of the 19^{th} century (Gabriel 1976).

The GIS analysis suggests similarly extensive fires in the pre-exclusion period (fig. 6). Over this 46-year period, 404,365 hectares (999,208 acres) burned across the BMWC, leading to a natural fire rotation period of approximately 46 years (table 1). The higher FRP estimate produced by Gabriel (1976) likely results from both a difference in methodology, as tree ring analysis can underestimate fire activity due to loss of fire scars in stand replacing fire events, as well as a difference in study extent. While Gabriel (1976) focused on the relatively cool and moist Danaher Drainage, this GIS analysis was conducted for the entirety of the BMWC.

As in the SBW, Native Americans likely influenced the fire regime in the BMWC. Ayres (1900) cites Native Americans as a source of ignition for the fires he observed and noted that signs of burning were frequent along commonly used trails and campsites. In addition, dating of barkpeeled trees suggests that Native Americans from the Salish, Kootenai, and Blackfoot tribes actively used the BMWC over the period 1665-1938, with use peaking from 1851-1875 (Östlund et al. 2005). Again, fire use was largely clustered along high-use campsites and travel corridors.

Fire Exclusion (1935-1973)

The BMWC saw a dramatic decrease in area burned following the implementation of the 10 A.M. policy in 1935 (fig. 6). The policy was successful in the BMWC due to the buildup of trails, telephone lines, and lookouts (Steele 1960). In part due to fire suppression efforts, combined with a cooler and wetter climate, only 2,080 hectares (5,140 acres) burned in the BMWC from 1935-1982, resulting in an FRP of 9,230 years (Table 1). Like the SBW, lightning ignitions remained common within the BMWC, with approximately 220 ignitions in the BMWC from 1940-1960 (Steele 1960), but 188 of those fires were kept to under a quarter of an acre, and only one grew to over 100 acres during these two decades (Steele 1960).

This exclusion of fire from the BMWC resulted in changes to forest structure in certain forest types. This was most dramatic on the east side of the Continental Divide, where fire exclusion resulted in encroachment of extensive grasslands by lodgepole and limber pine (Steele 1960). Fire exclusion also resulted in a decline in the number of ponderosa pine parklands along the South Fork of the Flathead (Arno et al. 1995; Keane et al. 2006; Steele 1960). Frequent, low severity fires, probably ignited by Native Americans, had maintained this forest type by burning through and reducing undergrowth density. With the exclusion of fire, lodgepole pine began to establish and increase the risk of crown fire reaching the older ponderosa pine species (Larson et al. 2013). Overall, homogeneity of forest structure increased, as the fire-driven stand openings and mosaics were replaced by larger, even-aged stands of shade-tolerant species (Arno et al. 2000; Steele 1960)

Transition period (1974-1982)

In 1974, two years after the beginning of the White Cap Study in the SBW, Orville Daniels migrated from the Bitterroot National Forest to take over as the Forest Supervisor for the Lolo National Forest. Building off the momentum and lessons from the White Cap Study, Daniels wrote a fire management plan for the Lolo National Forest, including the sections of the BMWC that fall within the Lolo National Forest. According to Daniels, writing this plan followed the same steps established by the White Cap Study, as the plan relied on extensive data collection and careful planning for each land management unit (retired U.S. Forest Service, Missoula, MT, personal communication, November 5, 2018).

The Lolo fire management plan was enacted in 1978, following the repeal of the 10 A.M. policy. In 1979, one fire was managed for resource benefit under this plan. It was, however, ultimately suppressed at a couple thousand acres due to concerns that it would affect sensitive hydrologic areas (Orville Daniels, retired U.S. Forest Service, Missoula, MT, personal communication, November 5, 2018).

The shift in national policy that allowed for management of wildfire for resource benefit in 1978 inspired land managers within the BMWC to begin working on fire management plans for the wilderness complex. In 1981, the Scapegoat-Danaher fire management plan was put into effect for the southern portion of the wilderness complex. Extensive ecological inventories conducted in 1979 and 1980 by Sonny Stiger, the fuels specialist for both the Helena and Lewis and Clark National Forests, laid the groundwork for this plan (Norman Kamrud, Retired U.S.

Forest Service, Missoula, MT, personal communication, October 18, 2018). Stiger and John Robertson, a fire staff officer for the Flathead National Forest, teamed up to write the majority of the Scapegoat-Danaher plan, which the four forest supervisors for this area signed off on in 1981 (Norman Kamrud, Retired U.S. Forest Service, Missoula, MT, personal communication, October 18, 2018).

Fire Management (1983-present)

In 1983 the Scapegoat-Danaher fire management plan was expanded to include the rest of the Bob Marshall and Great Bear Wildernesses, thereby allowing for wilderness fire management in the entirety of the BMWC (USDA Forest Service 1983). The summer of 1983 was also the first time a wilderness fire burned within the BMWC. The Cigarette Creek Fire ignited on the Lewis and Clark National Forest in late July and initially smoldered before rising temperatures increased fire activity in early August (Norman Kamrud, retired U.S. Forest Service, Missoula, MT, personal communication, October 18, 2018). Following detection, the fire received constant monitoring, both on the ground and in the air, which allowed the managers to continually predict potential fire activity and draft contingency plans. The Cigarette Creek Fire ultimately grew to 200 acres before it was extinguished by a fall rain event (Dave Bunnell, retired U.S. Forest Service, Missoula, MT, personal communication, October 10, 2018).

The next year, in 1984, the Lodgepole Fire ignited late in the season on the Flathead National Forest. Initially, the forest supervisor and district rangers were tempted to suppress the Lodgepole Fire because fire danger indices had been increasing and it had ignited near an airfield (Rich Lasko, retired U.S. Forest Service, Missoula, MT, personal communication, October 23, 2018). Nevertheless, the acting fire management officer (FMO) for the Flathead, Rich Lasko, advocated for managing the fire as a wilderness fire, and eventually was allowed to go forward without suppression action. According to Lasko, this largely entailed monitoring the fire from the Schafer Ranger Station next to the airfield (retired U.S. Forest Service, Missoula, MT, personal communication, October 23, 2018). It then burned as a slow-moving surface fire through a 1920s-origin lodgepole pine stand. The Lodgepole Fire ultimately grew to only 80 acres and was extinguished by rain events in the fall (Rich Lasko, retired U.S. Forest Service, Missoula, MT, personal communication, October 23, 2018).

The Lodgepole and Cigarette Creek fires were ultimately considered successful examples of wilderness fire management by both land managers and the public. Kamrud contends that these early fires allowed fire managers in the BMWC to better understand their individual roles and responsibilities within a long-term fire monitoring framework (retired U.S. Forest Service, Missoula, MT, personal communication, October 18, 2018). Furthermore, because both fires stayed within their MMA without any additional resources or funding, there were no serious complaints from the public or the larger land management community (Rich Lasko, retired U.S. Forest Service, Missoula, MT, personal communication, October 23, 2018). As the first wilderness fires in the BMWC, they proved that replicating the fire management approach used in the White Cap study area was a feasible strategy.

The most formative fire for early wilderness fire management in the BMWC, however, was the 1985 Charlotte Peak Fire on the Flathead National Forest. According to Dave Bunnell, who was the FMO for the Flathead at the time, this lightning-ignited fire began in late June in a forest stand at 7,000 feet (retired U.S. Forest Service, Missoula, MT, personal communication, October 10, 2018). The spring of 1985 was very dry, which allowed for such an early-season, high-elevation ignition. When fire behavior analysts were brought in, however, they determined that Charlotte Peak did not have a good chance of growing to much larger than a couple thousand acres (Dave Bunnell, retired U.S. Forest Service, Missoula, MT, personal communication, October 10, 2018). It was therefore determined that the Charlotte Peak Fire would be managed as a wilderness fire.

This decision to not suppress the Charlotte Peak Fire was controversial. Bunnell remembers how both the Regional and National offices believed that the decision was excessively risky given the high fire danger index values for the season (retired U.S. Forest Service, Missoula, MT, personal communication, October 10, 2018). As a result, the Assistant National Director of Fire and Aviation flew out from Washington, D.C., to inspect the situation. Supporting the decision to continue to allow the fire to burn, however, were Orville Daniels of the neighboring Lolo National Forest and Bob Mutch, then operating out of the Region 1 office as the Regional Fuels Specialist (Dave Bunnell, retired U.S. Forest Service, Missoula, MT, personal communication, October 10, 2018). Ultimately, backed by these experienced wilderness fire managers, Bunnell and other fire personnel on the Flathead were able to make the case that extinguishing the Charlotte Peak Fire was both incompatible with wilderness ethics and unsafe for firefighters, and they continued to manage it as a wilderness fire (Rich Lasko, retired U.S. Forest Service, Missoula, MT, personal communication, October 23, 2018).

After smoldering for the first few weeks of July, a frontal passage with high winds hit the Charlotte Peak Fire. These high winds, combined with low humidity, caused fire behavior to exceed the previous predictions (fig. 7). On July 23rd, the fire made a large run, burning 3,500 acres and producing a smoke column that was visible for many miles (Rich Lasko, retired U.S. Forest Service, Missoula, MT, personal communication, October 23, 2018). Bunnell and Mutch now became worried that the Charlotte Peak Fire would jump the South Fork of the Flathead, at which point fire patterns would become harder to predict and many more recreational users would be in danger (Smith 1986). This was avoided, however, by an early-season rain event during the first week of August, which extinguished the fire at 5,500 acres (Dave Bunnell, retired U.S. Forest Service, Missoula, MT, personal communication, October 10).

The size of the Charlotte Peak Fire demanded that fire managers of the BMWC adopt a new approach to fire management, as the focus shifted from managing the actual perimeter of the fire to managing the potential area the fire might encompass (Dave Bunnell, retired U.S. Forest Service, Missoula, MT, personal communication, October 10). Given the heavy recreational use of the South Fork Flathead River corridor, the greatest risk of fire in the area was public safety. As a result, managers on the Flathead developed thresholds of predicted fire danger or observed fire behavior, which, when exceeded, forced them to shut down trails to minimize the risk to wilderness users (Rich Lasko, retired U.S. Forest Service, Missoula, MT, personal communication, October 23). This required accurate long-range fire behavior predictions, which at the time meant qualified and experienced individuals using a TI-89 calculator to predict fire spread and movement (Dave Bunnell, retired U.S. Forest Service, Missoula, MT, personal communication, October 10). In addition, personnel were placed on the ground to monitor the fire and wilderness users in the area (Dave Bunnell, retired U.S. Forest Service, Missoula, MT, personal communication, October 10).

Despite the successes of these early fires, lack of funding and structural support consistently stymied wilderness fire management in the BMWC. Due to the continued emphasis on fire suppression, only $100,000 was allocated for wilderness fire management across all of Region 1, which was distributed on a first-come-first-serve basis (Bunnell, in preparation). This stood in stark contrast to fires managed as suppression events, which received nearly unlimited funding and resources. Because the SBW regularly had an earlier fire season than the BMWC, there was often no money remaining for even the first ignition in the BMWC. In addition, because wilderness fires did not fall under the same category as wildfires, they received very few additional outside resources (Bunnell, in preparation). All early wilderness fires within the BMWC were therefore managed completely by the local district or forest, often with a patchwork of funding and management strategies. This lack of funds and resources forced managers within the BMWC to suppress most fires due to lack of funds or personnel, regardless of the weather or fuels conditions (Bunnell, in preparation).

These strict limitations on wilderness fire funding loosened in 1988 when two large fires ignited within the BMWC. The Gates Park Fire and Canyon Creek Fire became defining events for the BMWC wilderness fire program, but for different reasons. The Gates Park Fire was eventually heralded as an emblem of successful wilderness fire management, whereas the Canyon Creek Fire crossed the wilderness boundary but provided important lessons on the inherent complexity and risk of wilderness fire management. Ultimately, the fires of 1988, both within the BMWC and across the Northern Rockies, fundamentally changed opinions on what could be done to affect fire perimeters and control fire behavior. It became clear to wilderness fire managers that there would be wilderness fires for which more active management of the fire, beyond just passive monitoring, would be required. Therefore, keeping wilderness fire management alive would involve increased investment in the program.

The Gates Park Fire ignited on the Lewis and Clark National Forest in July of 1988 (fig. 8). Norman Kamrud, the FMO for the Rocky Mountain ranger district, recalls initially using a small crew composed of trail crew members, recreation technicians, and fire managers from the district to suppress localized growth of the fire and keep it away from the nearby Gates Park administrative cabin (retired U.S. Forest Service, Missoula, MT, personal communication, October 18, 2018). Given the size and high levels of fire activity, however, Kamrud determined that keeping the fire within wilderness boundaries would require a larger, more specialized force.

Following Kamrud's recommendation, Dave Bunnell took over management of the Gates Park Fire. This change in command allowed for Kamrud to focus on the other fires within his district, while Bunnell was able to apply his considerable experience in wilderness fire management to this difficult fire event. Bunnell scraped together a small cooperative team from neighboring forests as well as the regional office (Dave Bunnell, retired U.S. Forest Service, Missoula, MT, personal communication, October 10, 2018). Bunnell lead this team for the duration of the Gates Park Fire, even after the regional budget allocated to wilderness fire ran out. At that point, Bunnell worked with the regional office and the forest supervisors of the Lewis and Clark, Lolo, and Helena National Forests to cobble together the resources to continue to manage this fire as a wilderness fire, rather than a suppression event (Dave Bunnell, retired U.S. Forest Service, Missoula, MT, personal communication, October 10, 2018).

In early August, Bunnell determined that the southern flank of the Gates Park Fire required direct action to keep the fire within wilderness boundaries (retired U.S. Forest Service, Missoula, MT, personal communication, October 10, 2018). He therefore called in a team of smokejumpers and hotshots to dig line and burn out this portion of the fire perimeter, which was the first large, direct action taken on a non-suppression wilderness fire (Bunnell, in preparation). These management actions, combined with a well-timed rain event and topographic features along the eastern flank, successfully contained the Gates Park Fire within the wilderness boundaries. The fire was extinguished naturally at the end of the season after burning 50,000 acres (Bunnell, in preparation).

In contrast to the Gates Park Fire, containment of the Canyon Creek Fire ultimately proved impossible, and the fire escaped the wilderness boundary (fig 8). This fire grew to over 200,000 acres and, following its escape, caused damage to private property and threatened the town of Augusta, Montana (Bunnel, in preparation). When the fire ignited on June 25th, a team of expert fire analysts predicted that total area burned would be limited to 3,000 acres, which seemed accurate throughout the early weeks of the fire (Orville Daniels, retired U.S. Forest Service, Missoula, MT, personal communication, November 5, 2018). Even with a two-day, 10,000-acre run starting on July 22, the Canyon Creek Fire remained within prescription and continued to burn with mixed fire behavior (Chaney 2013). These predictions, however, assumed that the typical seasonal rains would occur in mid- to late-August (Chaney 2013; Soulé and Knapp 2008). When August came and went with no such rain, the situation became highly unpredictable.

Further complicating management of the Canyon Creek Fire was the fact that 1988 was a big fire year for much of the Northern Rockies. At the same time Gates Park and Canyon Creek fire were burning in the BMWC, so too were fires in Yellowstone and Glacier National Parks, as well as several large fires along the Montana-Idaho border. Therefore, no resources remained to fight the now large Canyon Creek Fire, which had increased to 51,000 acres with dry wind events on August 29th (Chaney 2013). Then, the morning of September 6th, a low-level jet stream hit the fire, and in 16 hours the fire grew by another 180,000 acres. At this point, Daniels recounts, “there was nothing I could do” (Chaney 2013). Only a last-minute bulldozer line and some overgrazed pastures prevented the fire from burning into the town of Augusta, Montana (Orville Daniels, retired U.S. Forest Service, Missoula, MT, personal communication, November 5, 2018).

As the first fire to escape wilderness boundaries, many thought that the Canyon Creek Fire would mark the end of wilderness fire management. Instead, the Secretary of the Interior and Secretary of Agriculture convened a Fire Management Policy Review Team to evaluate the 1988 fires (van Wagtendok 2007). The team ultimately concluded that, although sound ecological premises supported the policy of wilderness fire management, not all areas using wilderness fire had adequate plans to guide decision-making (Wakimoto 1990). The team therefore recommended re-writing fire management plans, improving training for wilderness fire managers, and strengthening information dissemination to the media and general public (Wakimoto 1990).

The massive amounts of attention, from the media and politicians, that was given to the Yellowstone Fires of 1988 largely kept the Canyon Creek Fire and the BMWC out of the national spotlight. This lack of national blowback, in combination with the recommendations provided by the Fire Management Policy and Review team, ultimately allowed the BMWC and the Forest Service to learn and grow from the 1988 fires (van Wagtendok 2007).

Reflection on both fires revealed a “rhythm” to long-term fire management that the short-term actions taken on suppression fires had never exposed. According to Bunnell, of the 87 days that the Gates Park Fire burned, major growth occurred during only seven days of extreme weather (retired U.S. Forest Service, Missoula, MT, personal communication, October 10, 2018). On those days, the relative humidity, temperature, and wind conditions all aligned to contribute to high fire activity. At all other times, direct management on the perimeter to control fire spread was relatively safe. Ultimately, better understanding these controls of fire growth has allowed for more successful fire management outside of a suppression context.

The 1988 fire season was also the first to demonstrate that not all wilderness fires would be small, heterogeneous fires easily managed with observation alone (Dave Campbell, retired U.S. Forest Service, Hamilton, MT, personal communication, November 28, 2018; Chaney 2013). Upon reflection, it became clear that such large fire seasons, and large-fire events, are normal for the BMWC, as historical documents such as the Ayres report and newspaper accounts from the late 19th century contained stories of similarly large blazes (Chaney 2013). Around this same time, some of the first tree-ring records of fire history also indicated that these large fires of 1988 were consistent with the fire history in the Northern Rockies (Romme 1982).

With this new realization, wilderness fire managers determined that greater consideration of resources available would be crucial to making decisions on wilderness fire management (Orville Daniels, retired U.S. Forest Service, Missoula, MT, personal communication, November 5, 2018). These fires also pointed to the importance of considering drought indices, as 1988 came on the heels of three years of drought (USDA 2002a; Chaney 2013). Finally, the Canyon Creek and Gates Park fires highlighted the importance of communication with the public (Orville Daniels, retired U.S. Forest Service, Missoula, MT, personal communication, November 5, 2018). Although a lot of trust was lost with the neighboring communities, and especially with Augusta, admitting to the mistake and working with local ranchers to buy hay and rebuild fences helped to regain some publish trust and support for the program (Orville Daniels, retired U.S. Forest Service, Missoula, MT, personal communication, November 5, 2018).

In the long run, the 1988 fires ultimately put wilderness fire management on more solid footing. Although the 1989 federal review put the program on hold for several years, this allowed for time to review fire management plans and fix potential shortcomings (Orville Daniels, retired U.S. Forest Service, Missoula, MT, personal communication, November 5, 2018). According to Bunnell, fire management was also repositioned to more even footing with suppression fires within the Forest Service (retired U.S. Forest Service, Missoula, MT, personal communication, October 10, 2018). To start, the Region 1 office had allocated emergency funds for continued management of Gates Park and Canyon Creek fires, despite running out of funding for the wilderness fire program by mid-August. This, and the creation of a specific team to manage the Gates Park Fire, set a new precedent for wilderness fire (Dave Bunnell, retired U.S. Forest Service, Missoula, MT, personal communication, October 10, 2018). From this period on, the Forest Service repositioned itself as a leader in wilderness fire management, and the BMWC strengthened its fire prescriptions.

Fire activity in the BMWC was lower during the 1990s, largely due to cooler, wet fire seasons, which kept the fires that did ignite relatively small (Dave Bunnell, retired U.S. Forest Service, Missoula, MT, personal communication, October 10, 2018). Like the SBW, however, fire events occurring elsewhere had a profound impact on the BMWC, including the 1994 South Canyon incident in Colorado and the 1994 Howling Fire in neighboring Glacier National Park. While the Howling Fire was eventually heralded as a success and provided recommendations for the expansion of fire management for resource benefit, the death of the 14 firefighters during the South Canyon incident and the subsequent policy review limited the extent to which these recommendations could be applied within the BMWC during the remainder of the 1990s.

The BMWC further refined fire management plans following the 1995 national review. That same year, Bunnell teamed up with Tom Zimmerman, who had managed the Howling Fire, to better define the needs of wilderness fire management teams (Zimmerman and Bunnell 2000). Zimmerman would later create prescribed fire support crews, which developed into the modern-day wildland fire use modules. These crews provided personnel specifically trained in meeting wilderness and long-term fire management needs (Zimmerman et al. 2011).

In 2003, 41 fires burned 100,000 acres in the BMWC, marking a new watershed moment for the wilderness (Borrie et al. 2006). Seth Carbonari, who served as the FMO for the Spotted Bear Ranger District, notes that the longer fire seasons with more area burned, such as 2003, have forced wilderness fire managers in the BMWC to think on longer time scales (U.S. Forest Service, Hamilton, MT, personal communication, November 28, 2019). For example, fires in 2003 threatened a number of structures within the wilderness. Since it was a high-activity year outside of the wilderness as well, fire-fighting resources overall were scarce. As a result, management personnel within the BMWC were using stock to put point protection measures into place weeks ahead of time, and some cabins were wrapped in fire resistant material for months (Seth Carbonari, U.S. Forest Service, Hamilton, MT, personal communication, November 28, 2019). In addition, public safety considerations demanded that wilderness managers close many trailheads and trails within the BMWC (Borrie et al. 2006). Many of these measures have been repeated in later regional fire years , including 2007, 2015, and 2017 (fig. 10).

The 2003 fire season was also the first year that the BMWC conducted a prescribed burn within wilderness boundaries to support wilderness fire management. According to Mike Munoz, the District Ranger for the Rocky Mountain Ranger District, prescribed burns were conducted along the South Fork of the Sun River and involved burning 16,000 acres over the course of 2003, 2009, and 2011 (U.S. Forest Service, Choteau, MT, personal communication, December 7, 2018). Prior to the prescribed burns, very few wilderness fires had been allowed in this area because of the high risk of escape from the wilderness boundary. The goal of this project, therefore, was to provide a fuel break for fires and protect structures at risk along the wilderness boundary. Perhaps surprisingly, the community of Augusta was very supportive of this mitigation action. Munoz argues, however, that the motivation was there to reduce the risk of fire escape because the last three summers had included wildfires that nearly escaped wilderness boundaries (U.S. Forest Service, Choteau, MT, personal communication, December 7, 2018). Since this prescribed burn was completed in 2011, Munoz and his fire management team have been able to safely manage more lightning-ignited fires in the area.

The benefits of wilderness fire management have become more apparent in the BMWC with the increased frequency of large fire years in the past decade (Deb Mucklow-Starling, retired U.S. Forest Service, Kalispell, MT, personal communication, January 16, 2019). The 2003 season revealed the utility of previous burns when managing wilderness fire, as reduced flame lengths and decreased fire intensities were observed in the areas burning within the footprints of Charlotte Peak and the 2000 Lewis Creek II fires (USDA 2003). The fires from this year were also incorporated into the 2003 Guidebook on Wildland Fire Use for the BMWC, which identities the larger fires of that year as potential fuel breaks along the wilderness boundary (USDA 2003).

Munoz similarly identifies 2007 as an important year for "lessons learned" (U.S. Forest Service, Choteau, MT, personal communication, December 7, 2018.) During that season, three fires burned on the Rocky Mountain Ranger District, one of which was managed as a wilderness fire while the other two were managed as suppression events. Each of the suppression events was four times as expensive as the wilderness fire and ultimately, suppression actions proved ineffective in controlling fire extent. In addition, Munoz notes that as second and third entries become more common in his district, he sees increased public support for wilderness fire management. This is largely due to the increased accessibility of areas that experience repeat burns, as well as the reduced smoke levels in areas that have burned multiple times in the past several decades (Mike Munoz, U.S. Forest Service, Choteau, MT, personal communication, December 7, 2018).

Clear and open communication has repeatedly proven to be an essential component of wilderness fire management within the BMWC. This first became evident following the escape of the Canyon Creek Fire in 1988. Since then, the importance of communication has been highlighted frequently as large fire years become more common. For example, after a wilderness fire escaped the BMWC boundary in 2015 and forced evacuation of the town of Heart Butte, Munoz feared losing public support for wilderness fire management within that community (U.S. Forest Service, Choteau, MT, personal communication, December 7, 2018). Therefore, when the Crucifixion Creek Fire burned the same area again in 2017, he allocated more resources and personnel to the community. This included conducting weekly public meetings in Heart Butte, as well as designating a public affairs officer (PAO), in order to keep the town informed of all fire updates and management decisions. As a result, the community had greater trust in the decisions made by Munoz and his management team and an improved understanding that the two fire entries have made their community more resilient to future fires (Mike Munoz, U.S. Forest Service, Choteau, MT, personal communication, December 7, 2018).

In addition to communication with the public, communication across administrative boundaries has been equally important to the BMWC wilderness fire program. Once large fire years became more normal than surprising, the district rangers within the complex began setting up a weekly meeting to talk through current fires and discuss possible future events (Deb Mucklow-Starling, U.S. Forest Service, Kalispell, MT, personal communication, January 16, 2019). This phone meeting enabled the rangers to talk through decisions with their peers, share knowledge and resources, and plan for future ignitions (Deb Mucklow-Starling, U.S. Forest Service, Kalispell, MT, personal communication, January 16, 2019).

With the increased frequency of large fires in the BMWC, however, maintenance of trails and campsites has emerged as a major challenge to wilderness fire management. Maintaining access to the backcountry is essential for continued support of wilderness fire management, as support tends to dwindle when fire continually limits access to the wilderness area (Mike Munoz, U.S. Forest Service, Choteau, MT, personal communication, July 10, 2019). Funding for trail and campsite restoration following fire, however, is often hard to obtain. Additionally, these trails and campsites allow visitors to view the positive ecological effects of the fire (Deb Mucklow-Starling, U.S. Forest Service, Kalispell, MT, personal communication, January 16, 2019). Therefore, when districts are able to secure funding for trail work post-fire, making the decision to manage a wilderness fire often becomes easier.

Although a later adopter of wilderness fire management than the SBW, the BMWC nevertheless played a crucial role in developing policy and management strategies for long-term fire management, both within Region 1 and nationally. For example, from events such as the Gates Park and Canyon Creek fires, fire management personnel gained valuable lessons in how to manage large, long-term fire events, as well as the limitations of fire management under extreme fire weather conditions. Many of these managers later held positions at the regional and national offices, and therefore played a role in shaping national fire management policy and mentoring future fire managers. The high levels of uncertainty inherent in long-term fire management, combined with the often high levels of risk, makes such expertise crucial to sustaining and growing fire management for resource benefit.

Current conditions

In the past 34 years, nearly a third of the wilderness has burned once, with almost 7% burning multiple times (table 3 and fig. 6). Most of the area in the BMWC that has burned more than once in the past several decades are low-elevation, ponderosa pine stands, which has removed undergrowth and reduced fuel loads within these stands (fig. 11b, Larson et al. 2013; Flanary and Keane 2020). In mixed conifer-western larch stands, there has been a similar effect of fire on forest structure, creating a more open understory while the large, fire-resistant western larch trees persist following fire (Belote et al. 2015). There is a marked difference in forest structure between these burned stands and those that remain unburned, which tend to have high levels of ladder fuels and greater uniformity of vertical and horizontal forest structure (fig. 11a). In higher-elevation, subalpine forests, many stands have experienced a high-severity fire in recent decades, often resulting in near complete overstory mortality. In many of these stands, however, regeneration densities are high, particularly where lodgepole pine is present (fig. 11c).

The cumulative effect of fire in these various forest types has been high levels of landscape diversity, especially where short-intervals fires are common (fig. 12a,c). In the BMWC, the moderating effects of a previous fire on subsequent fire activity has also been evident, as fire activity is often reduced when a current burn encounters a previously burned area (fig. 12b). As a result, fire management has become easier over time, as the high landscape heterogeneity helps moderate fire intensity and rate of spread under certain climate conditions.

Frank Church River of no Return Wilderness

Pre-Exclusion (up through 1934)

Unlike the SBW and BMWC, the Forest Reserve for the area that now comprises the FCRNRW was not established until 1906. Since all Forest Reserves were converted to management under the Forest Service in 1907, no USGS survey was conducted for this area.

Therefore, there is no on-the-ground estimate of fire history for the FCRNRW. Nevertheless, a tree ring fire-scar study conducted in the ponderosa pine-Douglas-fir River Breaks zone of the Salmon River drainage indicates that, for these low-elevation forest types, the mean fire return interval (MFI) from c. 1647-1935 ranged from 10-48 years across nine sampled stands. The MFI for major fires (>1,000 acres) within the same Salmon River Breaks drainage was 41 years (Barrett 1984). The study also notes that stand-replacing fire within the River Breaks zone appears to be common on the north-facing, predominantly Douglas-fir dominated sites, with some stands aged to over 150 years old (Barrett 1984).

For the entirety of the FCRNRW, the GIS analysis conducted for this review indicates an FRP of 293 years, with 149,864 hectares (370,322 acres) of burned area within the FCRNRW from 1880-1934 (table 1, fig. 13). Prior to fire exclusion beginning in 1935, however, sheep grazing may have reduced fire frequency in the area in the early 1900s (Steele et al. 1981).

Before Euroamerican settlement, the Nez Perce and Shoshone tribes primarily occupied the FCRNRW (Cannaday 2016; Cochrell 1960). Although there was no ethnography conducted at the time, and no bark peeling analysis has been done for this area, it seems highly likely that Native Americans affected the fire history of the area, especially prior to the era of Euroamerican mining (Steele et al. 1986). This is especially true for the lower elevation sites and passes within the FCRNRW, where Native American land use would have been most concentrated.

Fire Exclusion (1935-1978)

The same tree ring study of the Salmon River Breaks zone was unable to calculate a meaningful MFI for the exclusion period of the FCRNRW, due to an absence of evidence for fire. Across all stands sampled, the MFI ranged from 57 to 61 years (Barrett 1984). This lack of fire was significantly outside the historical range of variation and resulted in predominantly

Douglas-fir regeneration on moist sites, while ponderosa pine regeneration still dominated on the very dry south-facing slopes (Barrett 1984). Even where ponderosa pine regeneration continued to dominate, however, litter and duff depths were much greater than what would be expected under historical fire frequencies (Barrett 1984).

The fire atlas data suggest that fire exclusion helped lower fire activity in the FCRNRW, although like the rest of the Northern Rockies, fire danger was generally lower over this period due to climate trends. In the five decades of the exclusion period, only 72,151 hectares (178,289 acres) burned (fig. 13), corresponding to an FRP of 635 years, more than twice as long as the pre-exclusion period (table 1).

Transition period (1979-1983)

Before wilderness designation of the FCRNRW in 1980 (Central Idaho Wilderness Act 1980), managed wildfires were allowed in portions of the Payette National Forest, which contains 33.5% of what is now the FCRNRW. Gene Benedict, who served as FMO for the Payette from 1979-2000, led the way in introducing wildland fire management for resource benefit to the forest. Benedict had a background in silviculture and ecology, and he strongly believed that fire was an integral part of the ecosystem (Gene Benedict, retired U.S. Forest Service, McCall, ID, personal communication, June 14, 2018). Benedict was determined to put the fire management lessons learned from the White Cap Study into practice on the Payette National Forest.

After the repeal of the 10 A.M. policy in 1978, Benedict collaborated with the supervisor of the Payette National Forest, Sonny LaSalle, to write a Forest Fire Action Plan (Benedict et al. 1991). This plan, which was enacted in 1979, allowed for appropriate suppression response in select areas of the Payette National Forest (Benedict et al. 1991). This meant that, although all fires in this area would still be considered “wildfires,” they would not all have to be managed with full suppression. Instead, depending on values at risk, risk to firefighters, and cost of suppression, a fire under the Fire Action Plan could be managed with flexible response options. The options included confinement with primarily natural barriers and minimal direct action, containment only at spots or areas of high risk, or the traditional approach of complete control (Benedict et al. 1991). The area included under this original plan was a high-elevation forest, where the historical fire regime was infrequent, high-severity fires that often remained small due to the sparse vegetation (Benedict et al. 1991).

Fire Management (1984-present)

Four years after Congress designated the FCRNRW in 1980, a fire management plan was developed for the entire wilderness area. Since the FCRNRW spanned two Forest Service regions and four national forests (fig. 2c), much collaboration was necessary to produce this plan. As a result, the authors attempted to have both regions and all forests “playing off the same script” (Bob Mutch, retired Forest Service, Missoula, MT, personal communication, November 19, 2018). The resulting plan allowed for wilderness fire within the FCRNRW and continued to allow for appropriate suppression response in remote non-wilderness areas such as those included in the Fire Action Plan on the Payette (Benedict et al. 1991).

The following year, in 1985, the benefits of appropriate suppression response on the Payette became evident (Benedict et al. 1991). That year, the Savage Creek Fire ignited outside the wilderness but was managed largely with the confinement strategy, which allowed for observation only. At the same time, portions of the fire that were threatening private lands and areas of high timber value were controlled with fire line (Benedict et al. 1991). This approach was estimated to have reduced firefighting costs by $2 million, reduced the risk to firefighters, and protected sensitive habitat types (Benedict et al. 1991).

The 1989 fire season was active on the Payette National Forest as well, which resulted in many fires that were managed as wilderness fires. According to LaSalle, one lightning storm that summer resulted in 244 fire starts, 17 of which were designated for management as wilderness fire events (retired U.S. Forest Service, Powell, ID, personal communication, June 21, 2018). Given that these ignitions were coming on the heels of the 1988 fires in Yellowstone and the BMWC, however, the Region 1 office was concerned that not suppressing these fires would threaten the wilderness fire program as a whole. To alleviate these fears, LaSalle flew some of the regional office employees over the wilderness so that they could observe first-hand the relatively low risk that these fires posed to the public. Once these regional office employees observed the patterns and effects of wilderness fire, as well as the vast area LaSalle was working with, support for wilderness fire in the FCRNRW grew (Sonny LaSalle, retired U.S. Forest Service, Powell, ID, personal communication, June 21, 2018).

Some important lessons were learned from the 1989 fire season. The large number of fires revealed to FCRNRW personnel that, during such a high activity year, not all fires could receive equal priority for the resource allocation necessary to manage a wilderness fire (Jack Kirkendall, retired U.S. Forest Service, Missoula, MT, personal communication, December 3, 2018). Instead, managers developed a priority setting process, which included factors to consider when determining which fires were best suited for management for resource benefit. This meant deciding which fires were most likely to be successfully suppressed and which fires were better suited to help reach goals related to ecological processes (Jack Kirkendall, retired U.S. Forest Service, Missoula, MT, personal communication, December 3, 2018). This decision-making process was written into future fire management plans so that management decisions were more straightforward and could be made by a greater number of personnel on the forest, rather than just the forest supervisor or FMO (Jack Kirkendall, retired U.S. Forest Service, Missoula, MT, personal communication, December 3, 2018).

The 1994 and 2000 fire seasons were also very active, not just for the FCRNRW but nationally. In 1994, three fires that each burned over 100,000 acres occurred within the FCRNRW (Gene Benedict, retired U.S. Forest Service, McCall, ID, personal communication, June 14, 2018). In 2000, the same year as the Bitterroot Valley Complex, the Burgdorf Junction Fire burned nearly 50,000 acres, most of which was in either in wilderness or roadless areas (Morrison et al. 2000). Due to severe fire activity nationwide, military troops were called in to help with the Burgdorf Junction Fire firefighting efforts. The high fire activity and rugged nature of the FCRNRW, however, made controlling the blaze impossible. Instead, fire management activity was largely limited to steering the fire away from values at risk and limiting threats to public safety (Booth 2000).

During high-activity fire seasons such as these, unwillingness at the regional and national level to commit to long-term management strategies required that nearly every fire be declared a suppression event. Resources available for suppression actions on fires in the FCRNRW, however, were severely limited (Gene Benedict, retired U.S. Forest Service, McCall, ID, personal communication, June 14, 2018). Therefore, regardless of the suppression designation, managers on the FCRNRW often made the decision to not commit any resources to these fires. Instead, those resources were held in reserve for other fires that posed an immediate threat to values outside of the wilderness (Sonny LaSalle, retired U.S. Forest Service, Powell, ID, personal communication, June 21, 2018).

Under the constraint of limited resources, fires that received a strong initial attack within the FCRNRW were often those that threatened inholdings within the wilderness (Gene Benedict, retired U.S. Forest Service, McCall, ID, personal communication, June 14, 2018). As of 2009, there were 61 private- or state-owned inholdings within the FCRNRW, as well as 24 airfields (USDA 2009). This large number of inholdings demands that, nearly every year, wilderness fire management decisions are made by balancing the values at risk inside the wilderness against the scarcity of resources available. According to Benedict, the highly qualified personnel on the Payette National Forest, as well as the support he enjoyed from neighboring forests towards the end of his career, made such wilderness fire management decisions easier because he could rely on their feedback and help (retired U.S. Forest Service, McCall, ID, personal communication, June 14, 2018).

Management of fires across forest and regional boundaries has been a continual challenge to wilderness fire management in the FCRNRW. The Salmon River forms the divide between Region 1 and Region 4 of the Forest Service inside the wilderness. In the early years of fire management, LaSalle and his forest staff often referred to this river as the "iron curtain" (retired U.S. Forest Service, Powell, ID, personal communication, June 21, 2018). Early on in the FCRNRW fire program, many resources and funds had to be channeled into suppressing fires at this boundary to prevent them from passing into Region 1, which was unwilling to accept any more fires beyond what it was already managing within the SBW and BMWC (Sonny LaSalle, retired U.S. Forest Service, Powell, ID, personal communication, June 21, 2018).

Within Region 4, the divides between individual forests could be equally challenging, as there were differing levels of comfort in managing long-term fire events between the forests (Jack Kirkendall, retired U.S. Forest Service, Missoula, MT, personal communication, December

3, 2018). As a result, fire management decisions in the FCRNRW were often made based off politics and interpersonal relationships as much as actual fire risk. In particular, boundaries between forests and regions were more of an issue when personalities clashed, especially in the early years of fire management when suppression was still firmly ingrained (Byron Bonney, Retired U.S. Forest Service, Missoula, MT, personal communication, November 27, 2018).

Over time, processes for managing fires across boundaries within the FCRNRW improved. At first, this was largely a result of increased acceptance of wilderness fire management and improved relations across these boundaries. For example, in 1992 Jack Kirkendall moved from his position as the district FMO on the Payette National Forest under Sonny LaSalle and Gene Benedict to take over as the Bitterroot National Forest FMO (Jack Kirkendall, Retired U.S. Forest Service, Missoula, MT, personal communication, December 3, 2018). Kirkendall's good relations with Benedict resulted in greater willingness to share resources and responsibility between the forests. This, combined with the increased acceptance of wilderness fire management within Regions 1 and 4, made managing fires across their shared boundary much easier (Jack Kirkendall, Retired U.S. Forest Service, Missoula, MT, personal communication, December 3, 2018).

As wilderness fire management became more widely accepted and interpersonal relationships improved, fire management guidebooks for the FCRNRW were amended to include more comprehensive risk assessment processes. These improved plans helped with management across boundaries, as more of the risk management process was laid out ahead of time (Jack Kirkendall, retired U.S. Forest Service, Missoula, MT, personal communication, December 3, 2018). As a result, there is currently a system in place to help with coordination of fire management across jurisdictional boundaries in the FCRNRW. According to Chuck Mark, the current Forest Supervisor for the Salmon-Challis National Forest, there are conversations related to long-term planning between the various forest and regional offices beginning on day one of a wilderness fire event (U.S. Forest Service, Salmon, ID, personal communication, November 26, 2018). These conversations will involve identifying potential values at risk, plans for mitigating these risks, and planning for management decisions beyond the initial go/no-go decision.

More recent large fire years highlight the challenges inherent with managing fire in a time of increasing fire extent and severity. The Mustang Fire of 2012, for example, was extinguished at 336,028 acres on November 5th after over three months of burning (USDA Forest Service 2013). Given that the Mustang Fire ignited in a year when the national office had recommended aggressive suppression of all fires (Seielstad 2015; USDA Forest Service 2012), the forest declared it a wildfire. Initial attack, however, was unsuccessful. As the fire grew and extreme fire behavior increased, the incident management teams (IMTs) largely limited suppression activities to fire activity outside the wilderness (Bob Mutch, Retired U.S. Forest Service, Missoula, Montana, November 19, 2019).

The size and duration of the Mustang Fire presented unique management challenges. For example, five different incident management teams (IMTs) managed the fire throughout its duration (USDA Forest Service 2013). The following year, a Forest Service review of the Mustang Fire concluded that the change-over in IMTs had some negative consequences, including failure to maintain a consistent management or public communication strategy (USDA Forest Service 2013). The review team contrasted this with the Halstead Fire, which burned the same summer in a remote area of the FCRNRW. One National Incident Management Organization (NIMO) team managed the Halstead Fire for 44 days straight, or half the duration of the fire, which minimized the number of team turnovers. This lack of turnover resulted in a more coherent approach to fire management as well as stronger relationships between the NIMO team, local land managers, and the public. The review ultimately concluded that such consistency improved management of the Halstead Fire (USDA Forest Service 2013).

Perhaps most controversial fire in recent years was the 2013 Gold Pan Fire. Shortly after it ignited, Dave Campbell, who was still serving as District Ranger for the West Fork of the Bitterroot National Forest, flew over the fire. Given its location, he and Stu Hoyt, the FMO of the Bitterroot National Forest, concluded that it would likely burn downhill to the Selway River, which would limit the size of the fire to around 15,000 acres within the FCRNRW (Blois 2017). The computer models supported this prediction, and Campbell made the decision to manage the Gold Pan Fire as a wilderness fire (Blois 2017).

The week following that original flight, however, saw unpredictable record-breaking temperatures and extremely low humidity (Dave Campbell, personal communication, Hamilton, MT, May 21, 2018). This resulted in extreme fire behavior, with the Gold Pan Fire ultimately jumping the Selway River and growing to over 40,000 acres (fig. 14, Blois 2017). As a result, it threatened several campgrounds and came very close to the historic Magruder Ranger Station. The area burned by the Haystack Fire of 2005, however, created a fuel break that slowed and redirected the Gold Pan Fire, ultimately sparing the ranger station (Campbell and Mutch 2016).

Whether or not fire management personnel considered the Gold Pan Fire to be a disaster or a success highlights the importance of personality in wilderness fire management. In retrospect, Campbell argues that it was a success, since no structures were lost and he made the appropriate call with the information he had at the time (retired U.S. Forest Service, personal communication, Hamilton, MT, May 21, 2018). Those comfortable with wilderness fire management tend to agree with Campbell, while those with a more conservative view of wilderness fire, typically see the Gold Pan Fire as a disaster.

The severe consequences and unpredictability of the record-breaking weather that caused the Gold Pan fire to jump the Selway also reinforced one of the major lessons learned from the Canyon Creek Fire of 1988. As was the case with Canyon Creek, extreme weather made accurate fire behavior predictions impossible. The Gold Pan Fire provides another excellent example of the impossibility of total risk avoidance when managing fire for resource benefit, even with the best possible plans and personnel in place (Dave Campbell, retired U.S. Forest Service, personal communication, Hamilton, MT, May 21, 2018).

With fire years such as 2013 becoming more common, one of the biggest challenges for contemporary wilderness fire managers within the FCRNRW is managing wilderness recreation around the fires (Chuck Mark, U.S. Forest Service, Salmon, ID, personal communication, November 26, 2018). This is especially true given the popularity of rafting on the Middle Fork Salmon and Main Salmon rivers, and the reliance of the local economy on these rivers being open and accessible to the public (George and Minor 2003). This became apparent with the 2016 Teepee Springs Fire, which burned across the Salmon River near Riggins, ID. This fire forced wilderness managers to shut the river down to the public and to evacuate rafters (Associated Press 2015). The public reaction to these fires was largely negative, especially among the rafting guides in the area (Chuck Mark, U.S. Forest Service, Salmon, ID, personal communication, November 26, 2018).

In addition to immediate impacts of fire on recreation users and tourism, however, there are also fire effects that linger well past the initial event. Like Mike Munoz in the BMWC, Chuck Mark identifies trail and campsite maintenance as a major challenge to wilderness fire management (U.S. Forest Service, Salmon, ID, personal communication, November 26, 2018). This came to a head in the FCRNRW in 2013, when the Idaho legislature passed a non-binding resolution that declared the FCRNRW a disaster area because of the lack of trail maintenance in areas affected by fire (Dvorak 2013). This resolution clearly highlights public frustration with the loss of wilderness access. This ultimately threatens public support not just for wilderness fire management, but wilderness management more broadly. For example, the 2017 Salmon-Challis Forest Plan revision included a proposal to designate new wilderness areas. Some public comments in response to this revision indicated a lack of support for these additional wilderness designations due to the inability of the forest to maintain trails in existing wilderness areas (Chuck Mark, U.S. Forest Service, Salmon, ID, personal communication, November 26, 2018).

Although the FCRNRW was the last of the three large wilderness areas in the Northern Rockies to adopt fire management, it has faced some unique challenges that make it an interesting case study in wilderness fire management. More so than either the SBW or the BMWC, fire managers in the FCRNRW have had to contend with high values at risk from inholdings within the wilderness, frequent fire management across administrative boundaries, and fire impacts on recreation. As a result, managers within the FCRNRW have built upon the lessons learned from the SBW and the BMWC to adapt to their unique circumstances. As larger, longer fire seasons occur more frequently, the challenges faced by the FCRNRW are likely to become a more common occurrence across the western United States.

Current conditions

Of the three wilderness areas considered here, the FCRNRW has experienced the most fire activity since the introduction of wilderness fire management in the mid 1980s. Only 22% of the wilderness remains unburned, 40% has been burned twice, and over 20% has burned three or more times (table 3). Some areas of the wilderness have even burned up to nine times, by far the highest frequency of fire in any of the wilderness areas covered in this review (fig. 6).

As a result, like in the SBW and the BMWC, many ponderosa pine stands are now characterized by the open, park-like forest structure that likely existed prior to the fire exclusion era (Lloret et al. 2011). Similarly, structural heterogeneity has been restored in many mixed-conifer forest stands (fig. 15a). In addition, across the wilderness, there has been a dramatic reduction in tree densities generally, often resulting in conversion (either temporary or permanent) to shrub or grass vegetation (fig. 15b & c). Despite the high fire activity within this wilderness, however, fire in the FCRNRW has created high levels of landscape heterogeneity. This is evident in every major watershed, including the Salmon River (fig. 16a & b) and Selway River watersheds (fig. 16c).

DISCUSSION

This review uses the existing literature combined with new interview data to recount the history of and the lessons learned from wilderness fire management in the Northern Rockies. The Northern Rockies region has, since 1910, had a large impact on shaping national fire policy and fire management decisions. Thanks in part to the change set in motion by the Northern Rockies wilderness fire experiments, the 2009 guidance for implementation of federal wildland fire management policy currently allows for high levels of flexibility around wildland fire management and includes specific language on managing fire for ecosystem health (USDA and USDI 2009).

Implementing this flexible, ecologically informed approach, however, continues to challenge many land and fire managers. This is in part due to the culture of fire suppression within federal land management agencies (Schultz et al. 2019; Steelman and McCaffrey 2011;

Thompson et al. 2018). The 2009 policy, however, has also had some unintended consequences that have potentially limited wilderness fire management. For example, removing the distinction between suppression wildfire and wildfire use also removed the incentive for land management agencies to plan for and record the effects of "good" wildfire (Seielstad 2015; Thompson et al. 2018). As a result, there is concern that many of the wildfires that have burned since the 2009 policy change do so only because they resisted suppression, and therefore do not represent the ecological role that fire plays under a range of biophysical conditions (Seielstad 2015).

This continual default to fire suppression has had some devastating consequences, both ecologically and on human populations (McWethy et al. 2019; Schoennagel et al. 2017). Attempting to keep fire largely removed from the landscape has resulted in increased fuel loading and more homogenous landscape structure in many ecosystems that historically burned once every few years to decades. As a result, the risk of large, high-severity wildfires has simultaneously increased in these landscapes (Covington and Moore 1994; Larson and Churchill 2012; Lydersen et al. 2013). Furthermore, since fire suppression is typically effective under all but the most extreme conditions, most area burns in fires that escape primary and secondary attack under extreme weather conditions. This combination increases the risk of high-intensity, high-severity fires, as well as the risk of true fire catastrophes. Given these consequences, there is a growing momentum around reforming of forest fire management to manage more fire for resource benefit (North et al. 2015; O'Connor et al. 2016). The lessons learned from wilderness fire management in the Northern Rockies provide valuable guidance to federal land managers on how to best implement such reforms.

Fire management lessons learned

The lessons learned from wilderness fire management in the Northern Rockies highlights the importance of an individual's commitment to including fire as a fundamental component of the wilderness ecosystems. The White Cap Study alone highlights this, as it was the vision and persistence of Bud Moore that launched that study and catalyzed the shift away from fire suppression (USDA 2002b). Following in Moore's footsteps, however, were the fire and land managers interviewed for this review, all of whom prioritized the role of fire on the wilderness landscape over the existing culture of fire suppression, often at risk to their own careers (Canton-Thompson et al. 2008; Stephens et al. 2016). This is often referred to as a strong "wilderness ethic," or a commitment to the "untrammeled" nature of wilderness areas highlighted in the Wilderness Act. Although a wilderness ethic is hard to foster, given that it is largely intangible, attempts to do so would undoubtedly strengthen land manager's willingness to manage for wilderness fire.

More tangible than the concept of a wilderness ethic is the importance of long-term planning to successful wilderness fire management. Again, this was highlighted early on with the White Cap Study on the SBW. The two years of surveying and planning prior to the introduction of fire on that landscape allowed for informed management decisions to be made following the early ignitions. This ultimately demonstrated to the larger land management community that wilderness fire management could be conducted in such a way that reduced the overall risk of negative impacts of fires on humans lives and infrastructure.

As larger, longer fire seasons become more common (Abatzoglou and Williams 2016, McKenzie and Littell 2017, Holden et al. 2018), the complexity of wilderness fire management continues to escalate as wildfire risk and scarcity of fire-fighting resources also increases (Calkin et al. 2015, Seielstad 2015). Under such conditions, long-term planning becomes increasingly important, as was highlighted with the 1989 fire season in the FCRNRW and the 2003 fire season in the BMWC. In both these examples, fire managers recognized that limited resources would demand determining priorities and allocating resources early in the fire season to avoid later shortages. Such planning has been relied upon repeatedly since, and is often carried out by highly trained and skilled personnel (Doane et al. 2006, Schultz et al. 2019). The heavy reliance on such personnel in the wilderness areas of the Northern Rockies suggests that the ability to prepare and implement these long-term plans is a skill set that should be highly emphasized and promoted within wilderness fire management.

Even with the best plans and personnel, however, there is a level of risk inherent to wilderness fire management (Thompson and Calkin 2011). This lesson was highlighted in each of the wilderness areas, as the Independence Fire in the SBW, the Canyon Creek Fire in the BMWC, and Gold Pan Fire in the FCRNRW all exceeded initial predictions of fire behavior and extent and threatened values at risk. In each case, an extreme weather event, challenging to predict, caused the extreme fire behavior. These fires therefore reinforce the assertion that a high level of risk acceptance is necessary in wilderness fire management. Risk acceptance would be more likely to increase if incentives were in place, or disincentives were removed, to prioritize wilderness fire management.

Although wilderness fire management is inherently risky, management of fire through suppression only does not lessen risk. Rather, suppression defers risk until extreme weather makes continued suppression impossible, at which point fires burn at higher intensities and severities (Arno and Brown 1991; Calkin et al. 2015). Therefore, incentivizing thoughtful fire management plans, such as the wilderness fire plans covered in this review, would actually reduce wildfire risk in the long run. To maintain public support for such plans, however, land management agencies must increase public resilience to extreme fire events, like the Canyon Creek and Gold Pan fires.

Increasing social-ecological resilience to wildfire is a challenge that has plagued fire management since the end of the 10 A.M. policy (Gunderson and Holling 2002, Walker et al. 2004, McWethy et al. 2019). There are, however, some lessons to be learned from wilderness fire on this topic. In particular, the decades of wilderness fire management in the Northern Rockies suggest that communication between land management agencies and the broader public is a key component to fostering social-ecological resilience to wildfire. For example, the escape of the Canyon Creek Fire from the BMWC severely tested public support for the wilderness fire program. A concentrated public outreach campaign that focused on mitigating the negative impacts and rebuilding trust, however, was able to restore some of that support and minimize opposition to the program. This model of communication continues to be used in the Northern Rockies wilderness areas today, as Mike Munoz demonstrated with his handling of the 2015 and 2017 fires that threatened the town of Heart Butte. These examples highlight the importance of transparent and open communication between wilderness fire managers and the public (Lachapelle and McCool 2012, Rasch and McCaffrey 2019).

Finally, these three wilderness areas highlight the importance of cooperation across administrative boundaries. This became clear immediately following the White Cap Study, as Bud Moore recommended strengthening inter-agency cooperation to increase the ability to manage wilderness fire (Moore 1974). Communication and cooperation, both across boundaries and between agencies, becomes increasingly important as fire seasons grow larger and longer (Meyer et al. 2015; North et al. 2015; Sneeuwjagt et al. 2013). Fire managers from all three wilderness areas describe relying heavily on meetings between districts or forests, such as the conversations that take place each year within the FCRNRW, to make fire management decisions. This increased cooperation and communication ensures that the decisions made regarding wilderness fire management are as well-informed and safe as possible.

Ecological lessons learned

Beyond the fire management lessons learned, wilderness areas that have managed fire for resource benefit also allow researchers to investigate fire and forest ecology questions that cannot be answered elsewhere (Agee 2000). The presence of fire on the landscape in areas like the SBW, BMWC, and FCRNRW has allowed for research into the stand-level effect of fire on forest structure and fuel loads in a variety of forest types (e.g., Belote et al. 2015; Cansler et al. 2018; Kipfmueller and Kupfer 2005, Keeling et al. 2006, Larson et al. 2013). At the landscape-scale, wilderness fire has revealed a restoration of vegetation mosaics (Arno et al. 2000, Barrett et al. 1991). Finally, wilderness fire has also helped us learn how fire can operate as a self-regulating process, with previously burned areas moderating both the size and severity of subsequent fires (Parks et al. 2014, 2015d; Teske et al. 2012).

There has been abundant research in the Northern Rockies on the effects of natural fire patterns on other ecosystem processes. Research into soil nutrient cycling in frequently burned ponderosa pine-Douglas-fir stands in the SBW and FCRNRW revealed an increase in inorganic nitrogen availability following fire, which provided the first field-based evidence of decreased nitrogen availability resulting from fire suppression (DeLuca and Sala 2006). Similarly, within the BMWC, short-interval fire was found to significantly increase charcoal on coarse woody debris, which suggests that short-interval fires may alter carbon storage in forested ecosystems (Ward et al. 2017). Outside of the Northern Rockies, wilderness fire has allowed for investigation of fire patterns on forest hydrology and soil moisture levels (Boisramé et al. 2017, 2018). The presence of wilderness fires therefore provides laboratories of natural fire, which allows for research that can answer basic ecological questions and guide forest management outside of wilderness areas (Larson et al. *in press*).

CONCLUSION

Wilderness fire management has allowed for the continuation and restoration of fire as a fundamental ecosystem process within the large wilderness areas of the Northern Rockies. This has occurred in large part due to the hard work and courage of a few land managers who committed to the risk and learning curve of re-introducing fire, when the dominant cultural paradigm strongly supported fire suppression. Once fire was reintroduced, each subsequent wildfire or fire season provided new insight into wilderness fire management approaches and techniques. This increased management capability, combined with the increased resilience of wilderness ecosystems following the reintroduction of fire, has made wildfire management for resource benefit within the Northern Rockies more feasible over time.

Increased use of wildfire management for resource benefit will be crucial to solving the current fire management crisis that plagues the West (Barros 2018; North 2012, 2015). This is especially true for the large roadless areas, which, while not designated wilderness, are remote enough to make fuel-reduction strategies such as thinning or prescribed burning logistically and financially impossible (North 2014). Managing fire for resource benefit, however, can be difficult for managers, given the immediate risks involved, and the continued culture of suppression that pervades land management agencies (Schultz et al. 2019; Steelman and McCaffrey 2011; Thompson et al. 2018). Therefore, increasing wildland fire management for resource benefit requires land agencies to dramatically change how they approach wildfire management.

The practice of wilderness fire management in the Northern Rockies represents a relatively unique approach to fire management in the West. This long and vibrant history offers some important lessons for land managers seeking strategies other than fire suppression. By providing these lessons and setting the stage for a cultural shift in how the nation regards fire management for resource benefit, the Northern Rockies can once again play a pivotal role in shaping national fire policy and management.

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TABLES

Table 1. Area burned and fire rotation period (number of years required to burn an area equivalent to the total wilderness area) for the three wilderness areas of interest, according to the fire perimeter data available from pre-existing fire atlases.

	Pre-Exclusion		Exclusion		Fire Management	
	Area burned (ha)	FRP (yrs.)	Area burned (ha)	FRP (yrs.)	Area burned (ha)	FRP (yrs.)
Selway-Bitterroot Wilderness	443,213	55	10,439	2,194	577,670	37
Bob Marshall Wilderness Complex	404,365	46	2,080	9,230	472,857	29
Frank Church-River of No Return Wilderness	149,864	293	72,151	635	2,680,143	11

Table 2. Reproduced, with permission, from Mutch (1974). Prescription for each ecological land unit (ELU) within the White Cap Study Area as it was developed in 1972. BUI refers to buildup index, a fire danger metric used to measure fuel dryness.

ELU	Description (from Yurich 1976)	Suppression	Observation	Observation + Suppression
Shrubfield	-Warm, dry ecosystem -Composed of shrub vegetation and herbaceous understories -Results from repeated, short interval, high intensity fires.	a. Hunting season: BUI > 170 b. Along study boundaries	a. Prehunting season b. Hunting season: BUI < 170 c. BUI < 170	a. Fires approaching Wapiti Creek Range
Ponderosa Pine Savanna	-Hot, dry ecosystem -Low elevations, most commonly on steep south and southwest aspects -PIPO is the dominant, climax tree species with an understory composed of grass, forb, and shrub species. -Most flammable unit: low intensity, high frequency regime.		a. BUI < 170	a. BUI > 170
Ponderosa Pine/Douglas Fir	-Warm, dry ecosystem -South facing slopes -PIPO tends to be skewed to older aged trees, whereas PSME trees tend to be found in all age classes. -Open and diverse shrub layer	a. < 4500' elevation	a. > 4,500' elevation, BUI < 170	a. > 4,500' elevation, BUI > 170
North slope	-Warm, wet ecosystem -North facing slopes up to 6,000 feet -Composed of very diverse vegetation communities and relatively continuous cover -High fuel loadings; is defined by a low frequency, stand replacement fire regime.	a. Along study boundaries b. BUI > 170 at Peach Creek Drainage	a. West of Peach Creek Drainage b. Upper White Cap unit	a. BUI > 170 when fires approaching Peach Creek buffer
Subalpine	-Cold, wet ecosystem -Dominated by alpine fir and Engelmann spruce, with whitebark pine and alpine larch also present. -Hosts a low frequency fire regime, fires will range from low intensity spot fires to high intensity, stand replacing fires	a. Along study boundaries b. BUI > 170 at Bitterroot Crest Passes	a. Season-long	a. BUI < 170 when fires approaching Bitterroot Crest Passes

Table 3. Number-of-burns analysis for the SBW, BMWC, and FCRNRW during the fire management period.

	SBW 1978-2017		BMWC 1983-2017		FCRNRW 1984-2017	
Times Burned	Hectares burned	Percent of wilderness area	Hectares burned	Percent of wilderness area	Hectares burned	Percent of wilderness area
0	301,679	55	404,674	62	211,582	22
1	180,102	33	206,035	32	148,549	16
2	55,027	10	37,806	6	386,267	40
3	7,885	1	3,755	1	123,389	13
4	892	0.16	62	0.01	64,378	7
≥ 5	31	0.01	-	-	15,859	2

FIGURES

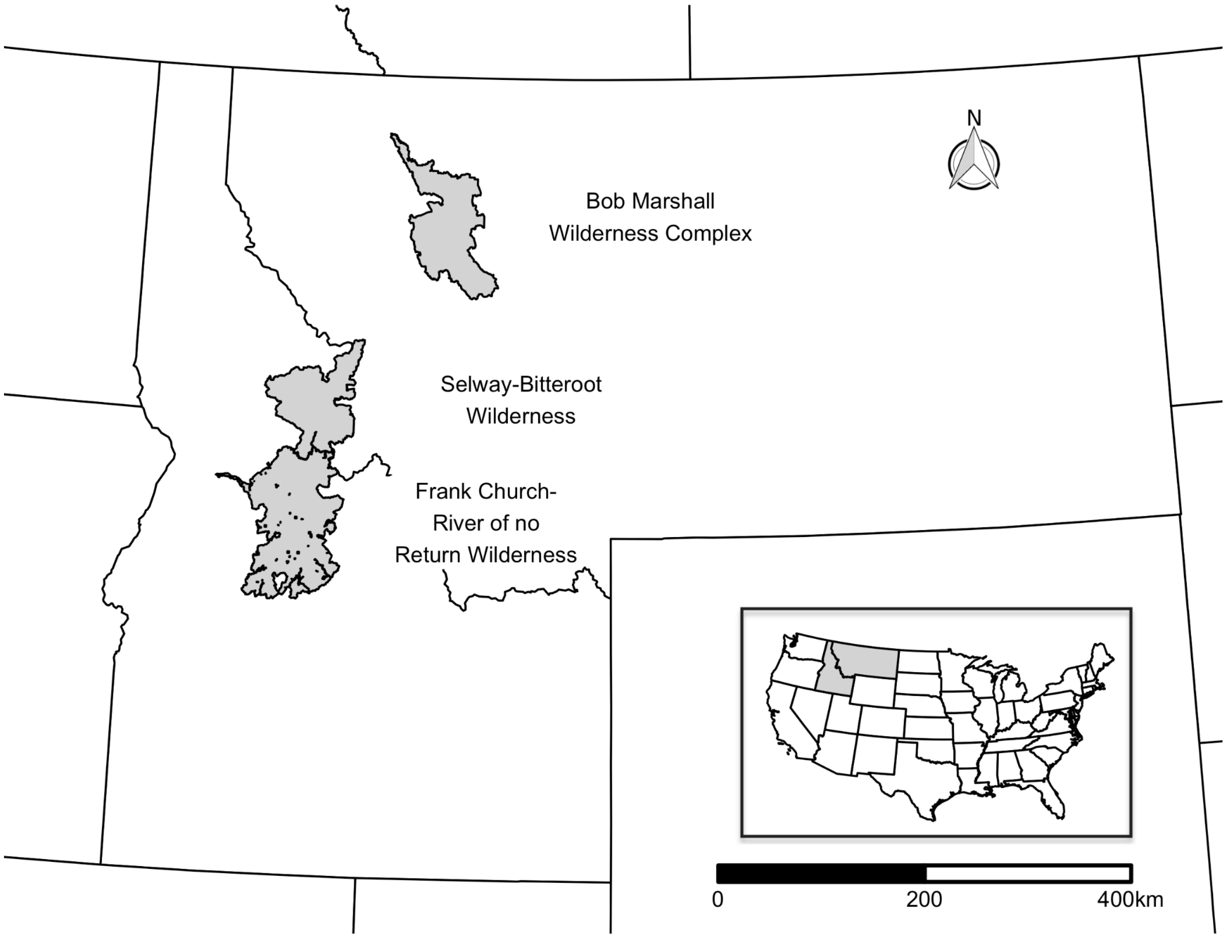

Figure 1. Wilderness areas covered in this review.

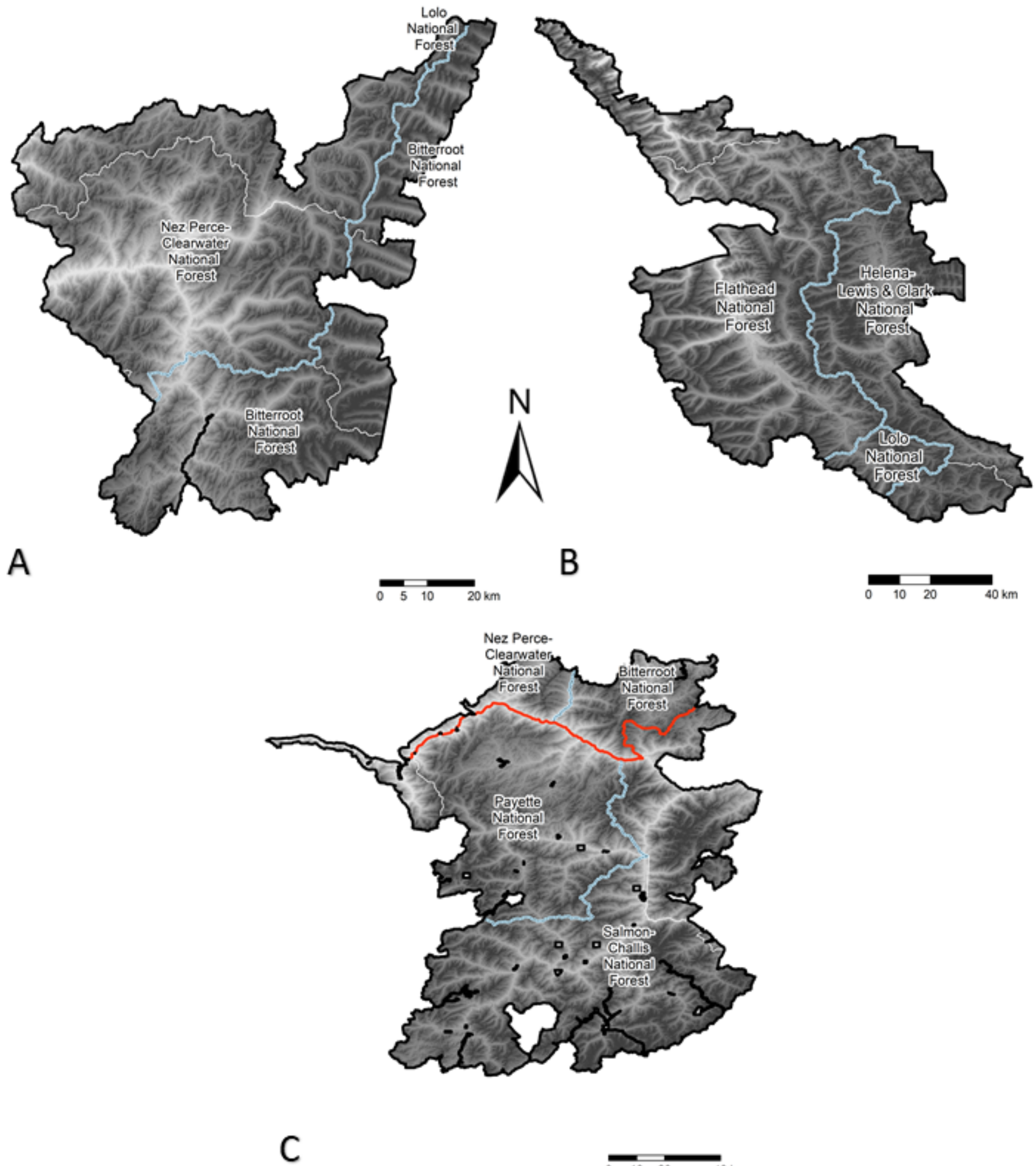

Figure 2. Administrative boundaries for the a) Selway Bitterroot Wilderness, b) Bob Marshall Wilderness Complex, and c) Frank Church-River of No Return Wilderness. Light blue lines indicate a national forest boundary, whereas thin white lines indicate a ranger district boundary. For the Frank Church, the boundary between Regions 1 to the north and Region 4 to the south is indicated by a red line.

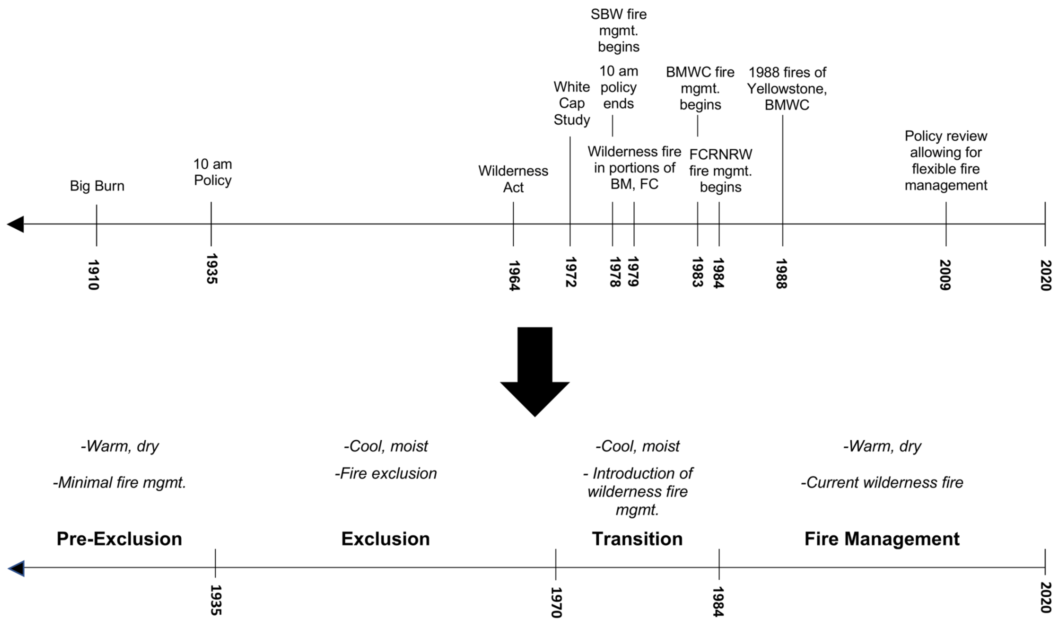

Figure 3. Timeline showing major fire events of the Northern Rockies, as well as shifts in federal policy and forest land management plans that impacted wilderness fire management in the Selway Bitterroot Wilderness, Bob Marshall Wilderness Complex, and Frank Church River of No Return Wilderness. These events lead to four major fire management eras: pre-exclusion, exclusion, transition, and wilderness fire management. The timing of the shift between exclusion and fire management varies in each wilderness depending on when a wilderness fire management plan was drafted for the entire wilderness.

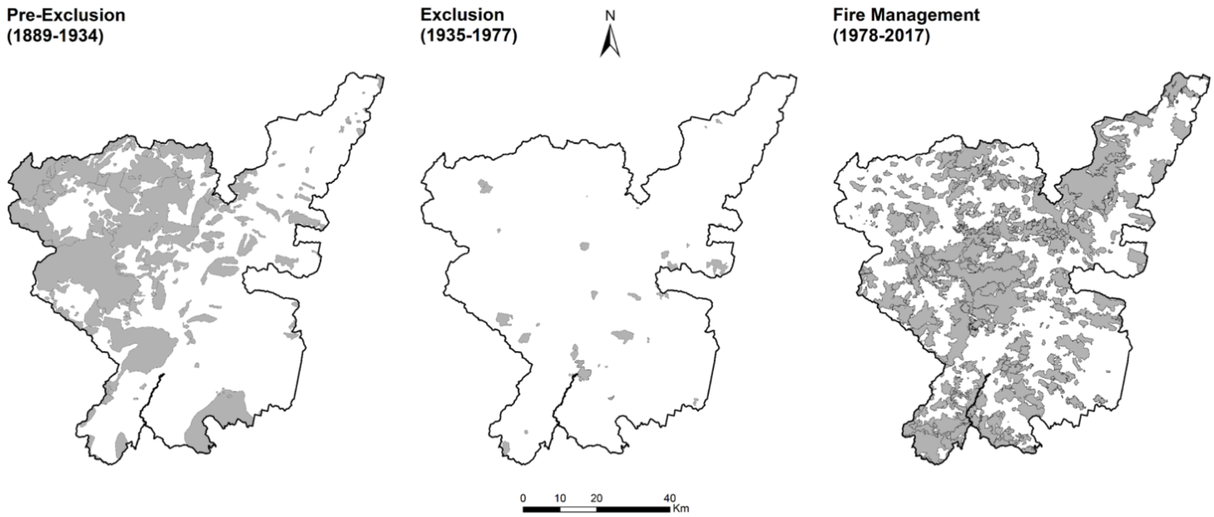

Figure 4. Fire perimeters for the three fire management periods in the Selway Bitterroot Wilderness of northwest Montana and eastern Idaho.

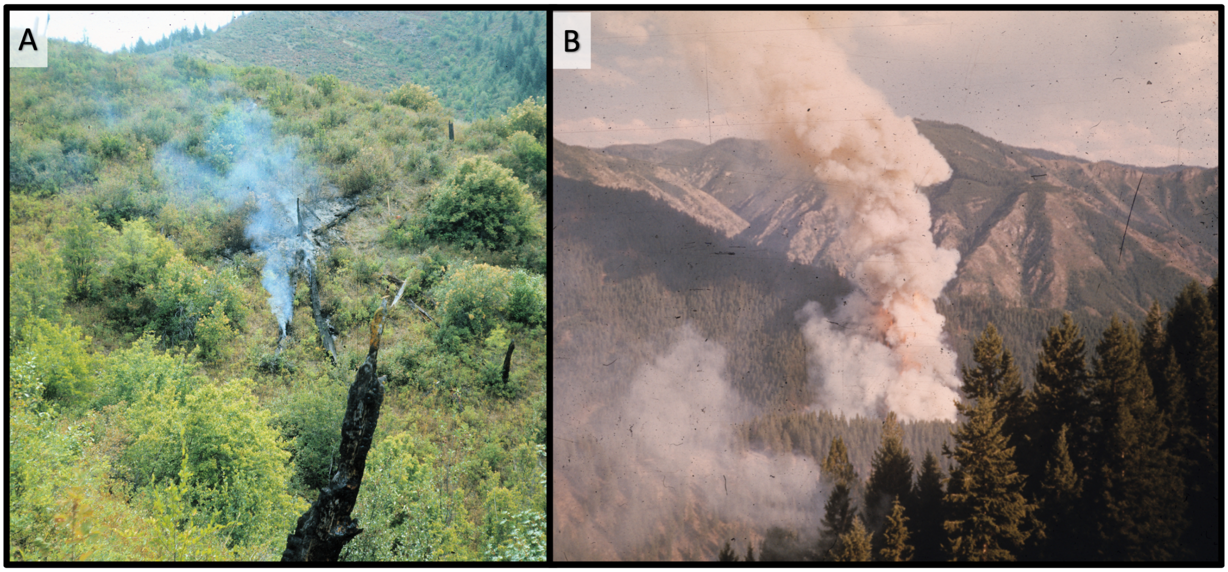

Figure 5. Early wilderness fires of the Selway Bitterroot Wilderness, including a) the Bad Luck Fire and b) the Fitz Creek Fire. Photography by Bob Mutch, retired U.S. Forest Service.

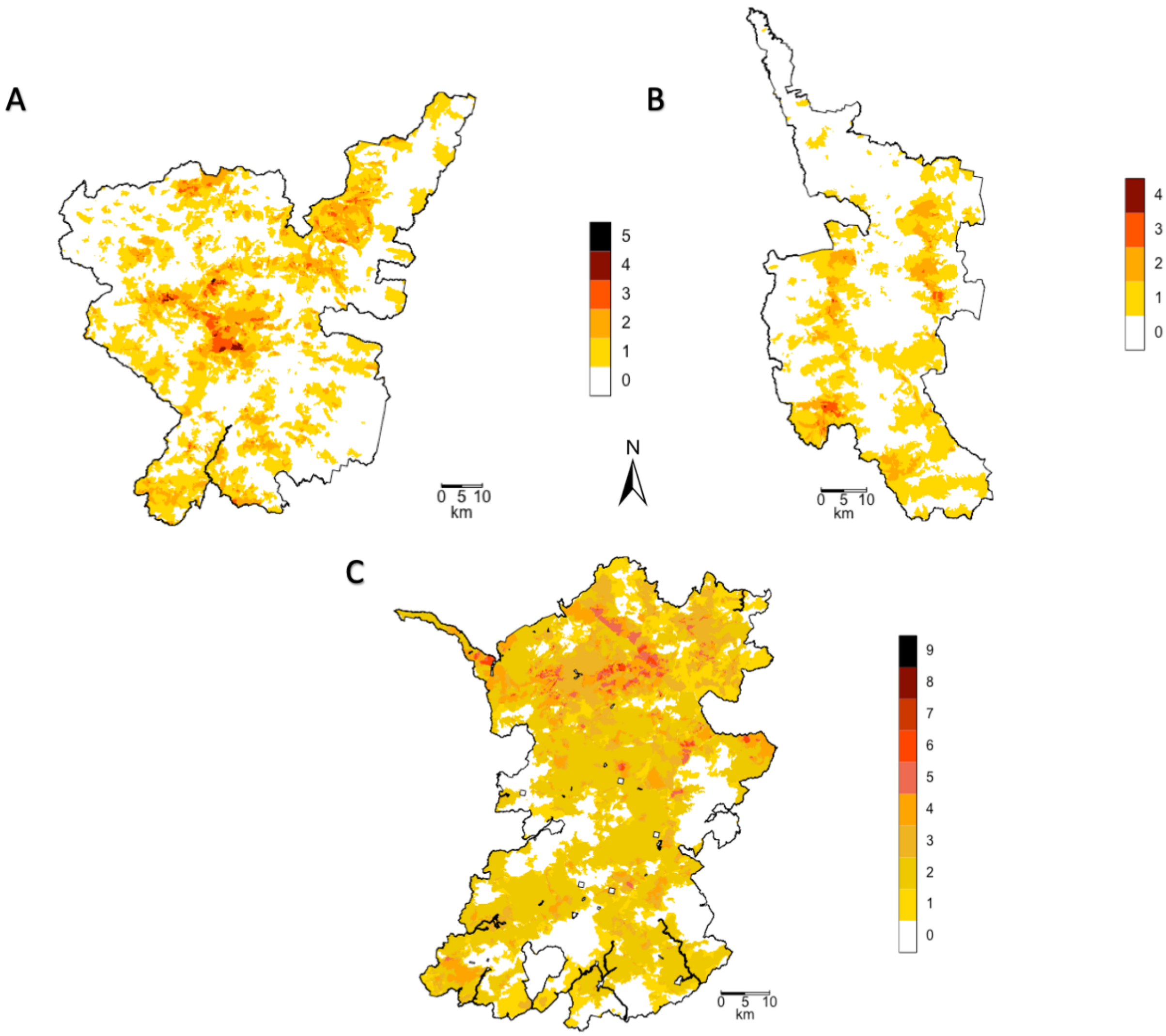

Figure 6. Number of burns on the landscape for a) the Selway Bitterroot Wilderness of Montana and Idaho from 1978-2017, b) the Bob Marshall Wilderness Complex of northwestern Montana from 1983-2017, and c) the Frank Church-River of No Return Wilderness of Idaho from 1984-2017.

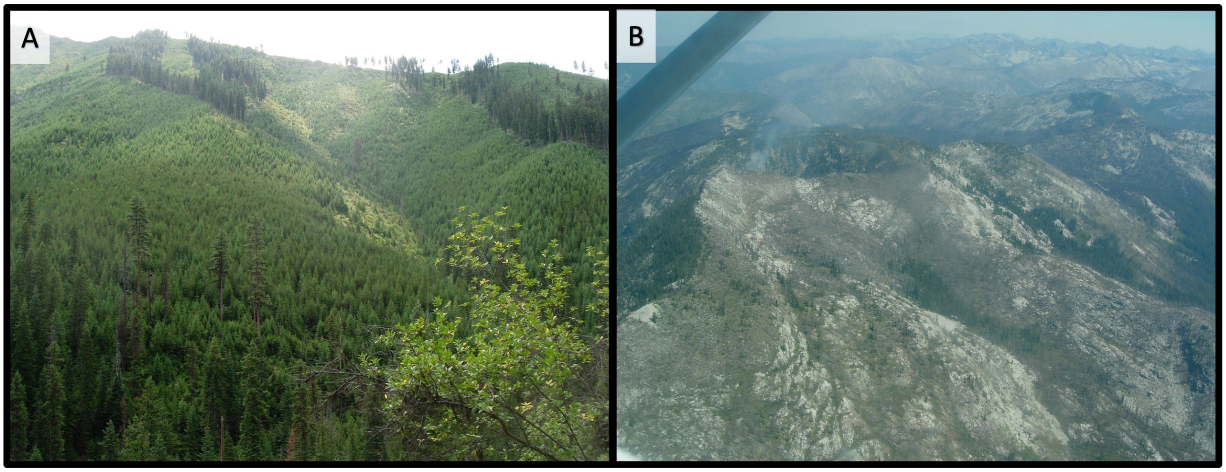

Figure 7. a) The forest structural diversity resulting from five decades of wilderness fire management is readily apparent within the White Cap drainage of the Selway Bitterroot Wilderness, where wilderness fire management within the Forest Service was born. b) A birds-eye-view of the landscape mosaics resulting from wilderness fire management in the Selway Bitterroot Wilderness, taken during the active fire season of 2007. Both photographs taken by Carol Miller, U.S. Forest Service.

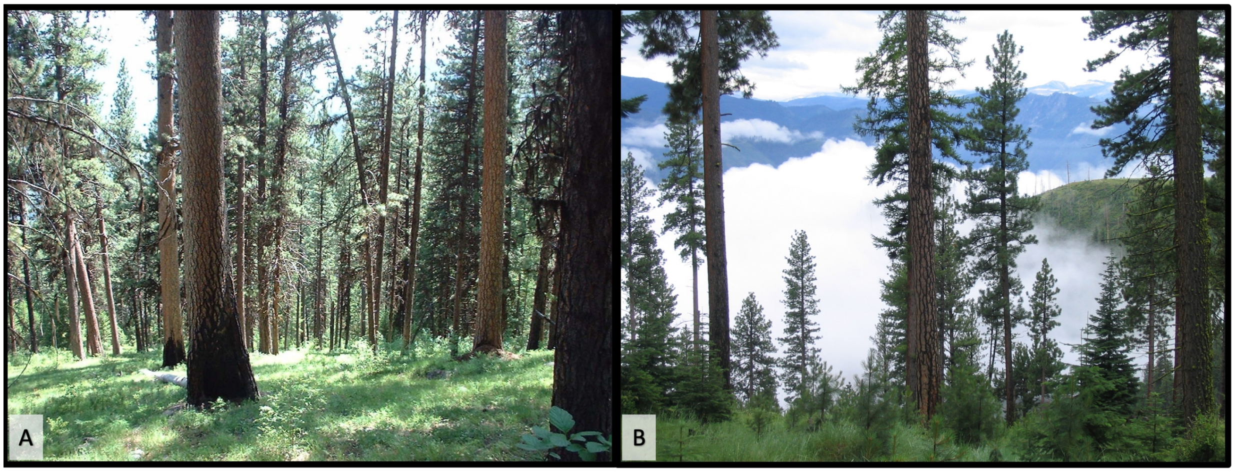

Figure 8. The stand structure resulting from a resumed fire regime in a) ponderosa pine and b) mixed-conifer, ponderosa pine-western larch forests of the Selway Bitterroot Wilderness. Photographs taken by Anna Sala, University of Montana.

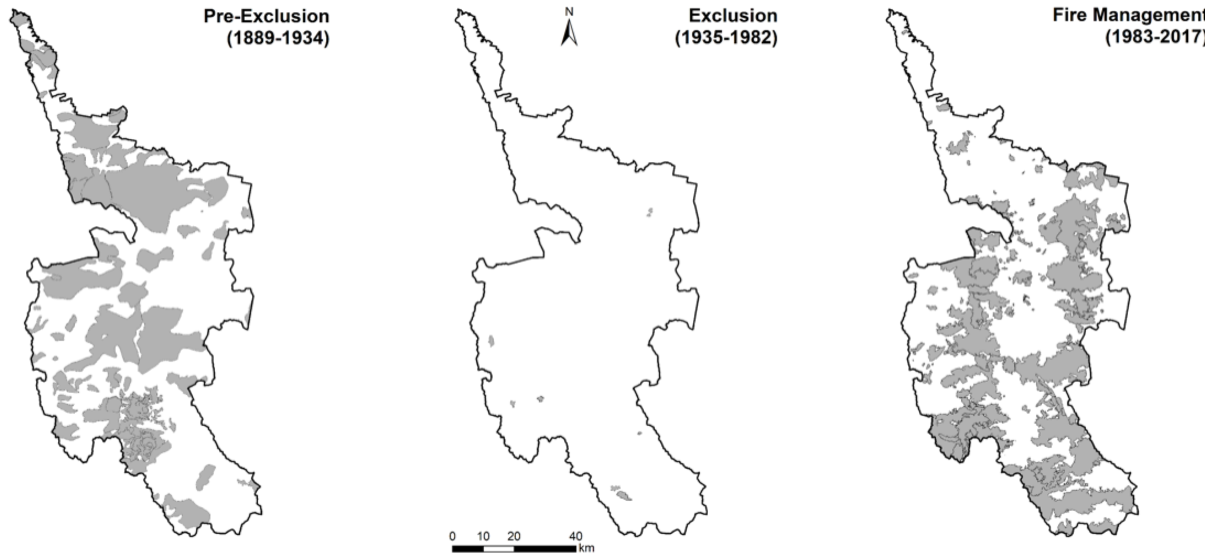

Figure 9. Map of fire perimeters for the three fire management periods of the Bob Marshall Wilderness Complex in northwest Montana.

Figure 10. A field crew watches the 2013 Damnation Fire burn in the Bob Marshall Wilderness Complex of northwest Montana. Photograph taken by Lily Clarke, University of Montana.

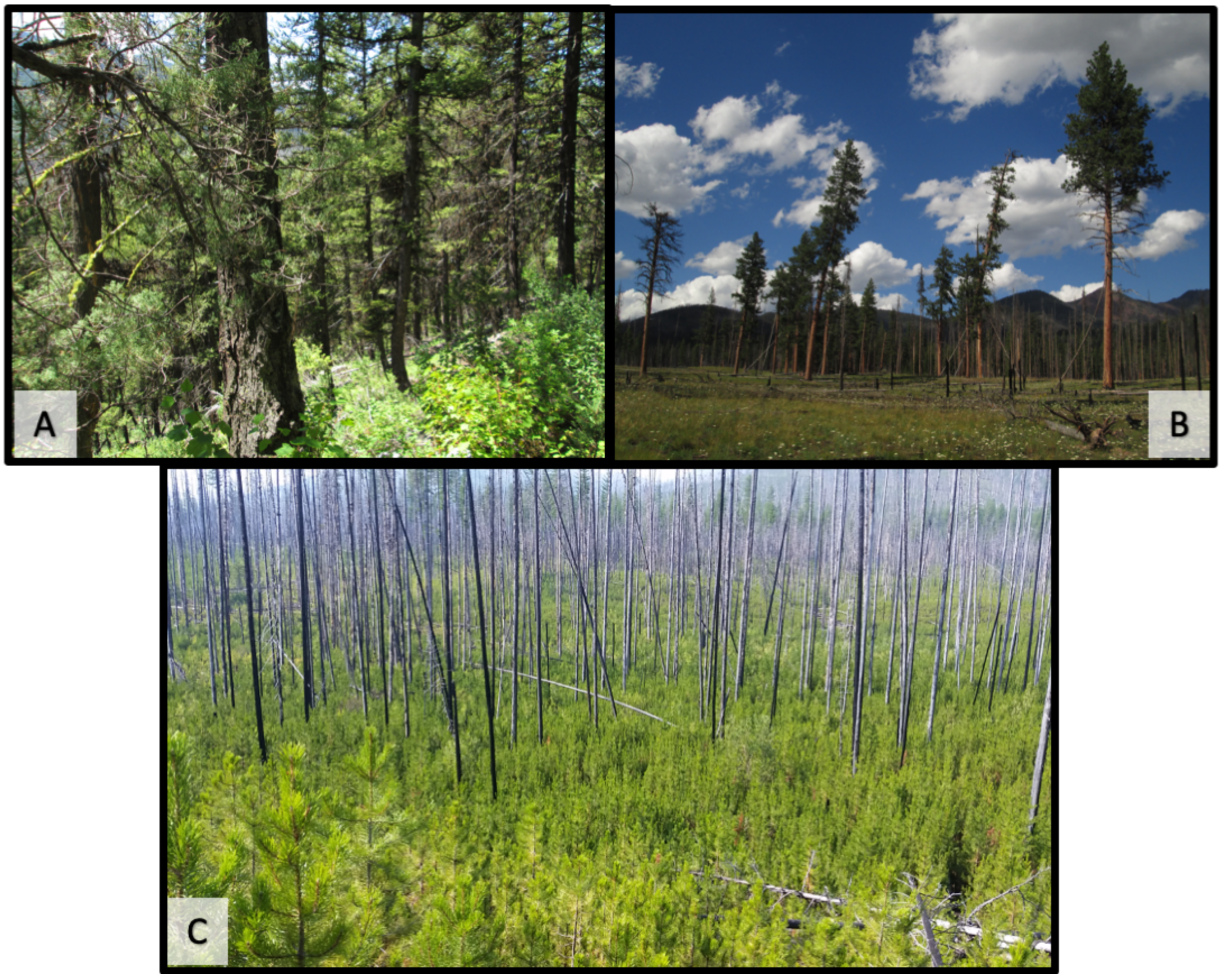

Figure 11. Stand structure in a) an unburned mixed conifer stand, b) a twice-burned ponderosa pine stand, and c) a lodgepole pine stand burned at high severity within the Bob Marshall Wilderness of northwest Montana. Photograph (a) taken by Eryn Schneider, University of Montana and photographs (b) and (c) taken by Andrew Larson, University of Montana.

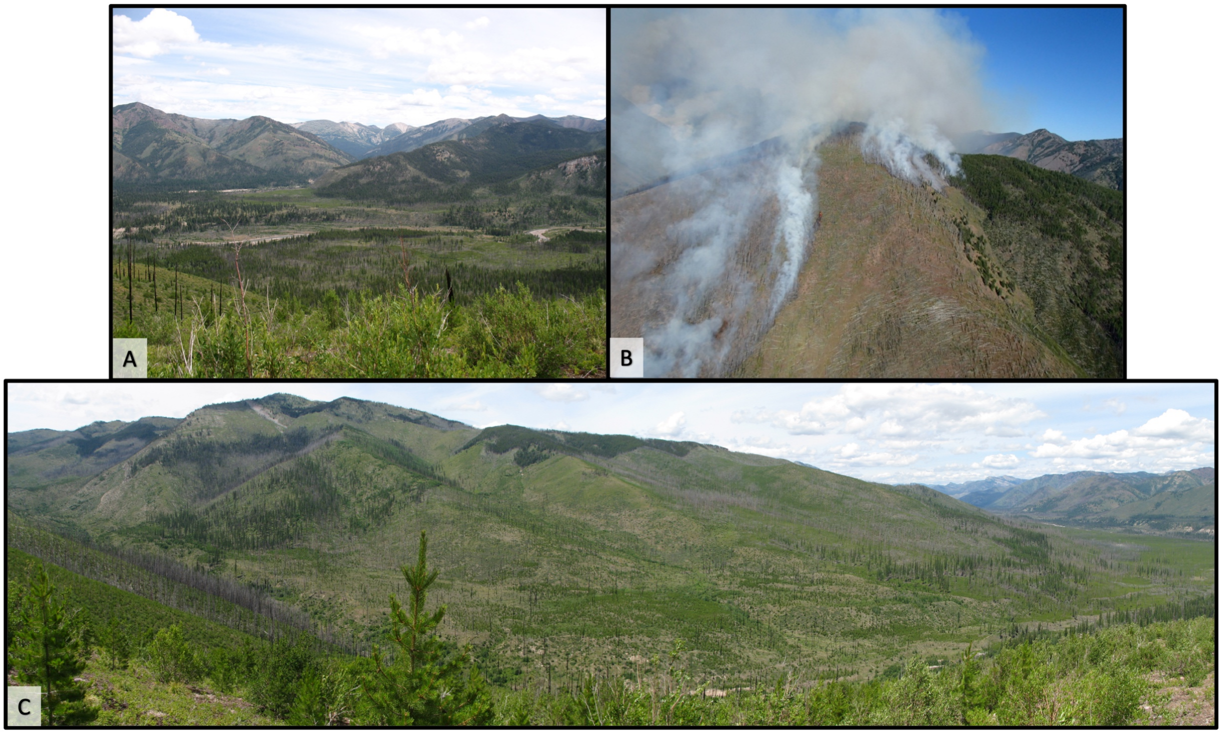

Figure 12. The landscape mosaics resulting from wilderness fire management in the Bob Marshall Wilderness of northwest Montana. a) Considerable structural diversity is evident at the confluence of the White River and South Fork of the Flathead River. Photograph by Julia Berkey, University of Montana. b) The 2013 Damnation fire burns through the mosaic left from the 2003 Little Salmon Complex at Mud Lake Mountain. Photograph by Seth Carbonari, U.S. Forest Service. c) Current-day forest structure within the perimeter of the 1985 Charlotte Peak fire. Photograph by Andrew Larson, University of Montana.

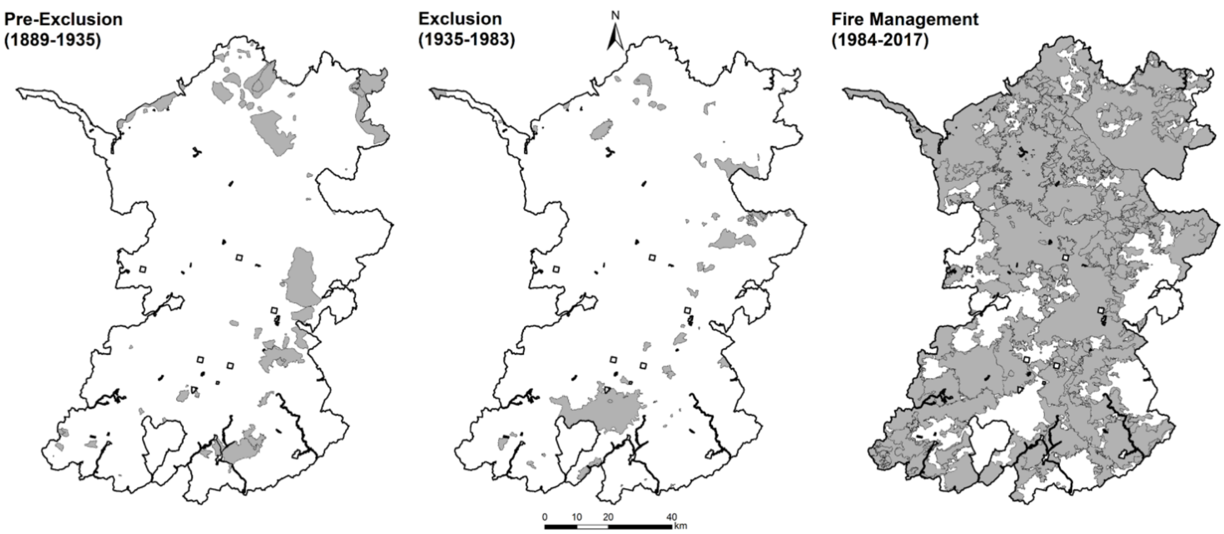

Figure 13. Map of the fire perimeters for the three fire management periods in the Frank Church-River of No Return Wilderness of eastern Idaho.

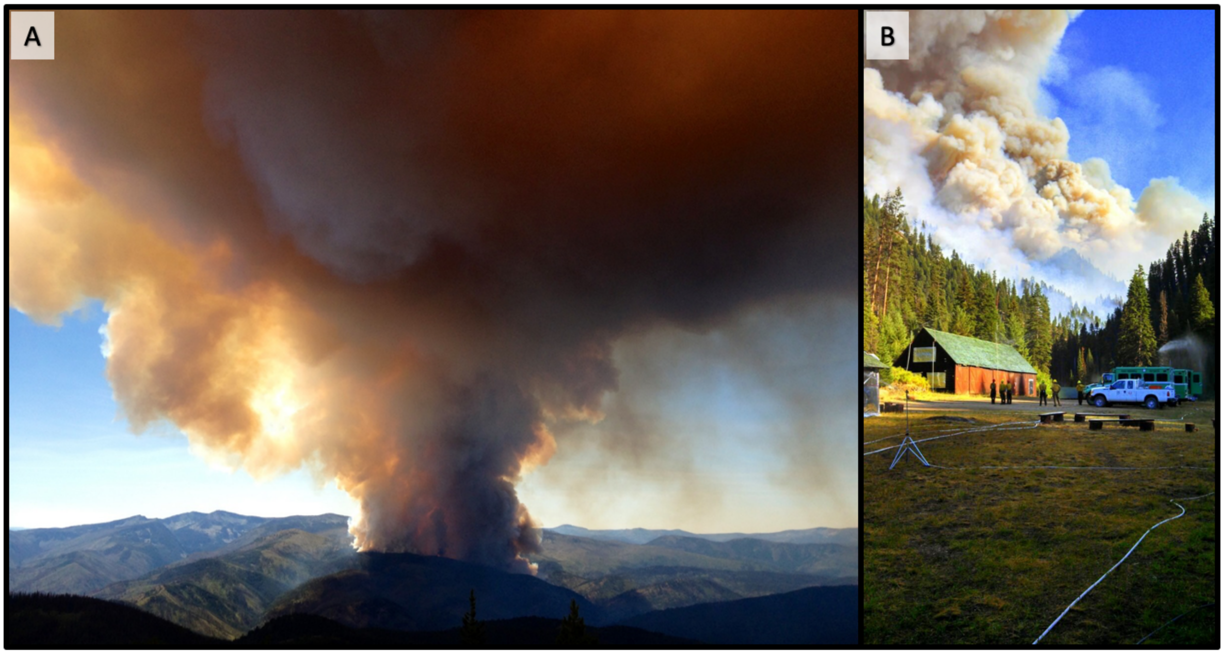

Figure 14. The 2013 Gold Pan fire in the Frank Church-River of No Return Wilderness in eastern Idaho, as viewed from a) Hells Half Acre Lookout and b) Magruder Ranger Station. Photographs taken by Mark Moak and Leah Moak, respectively.

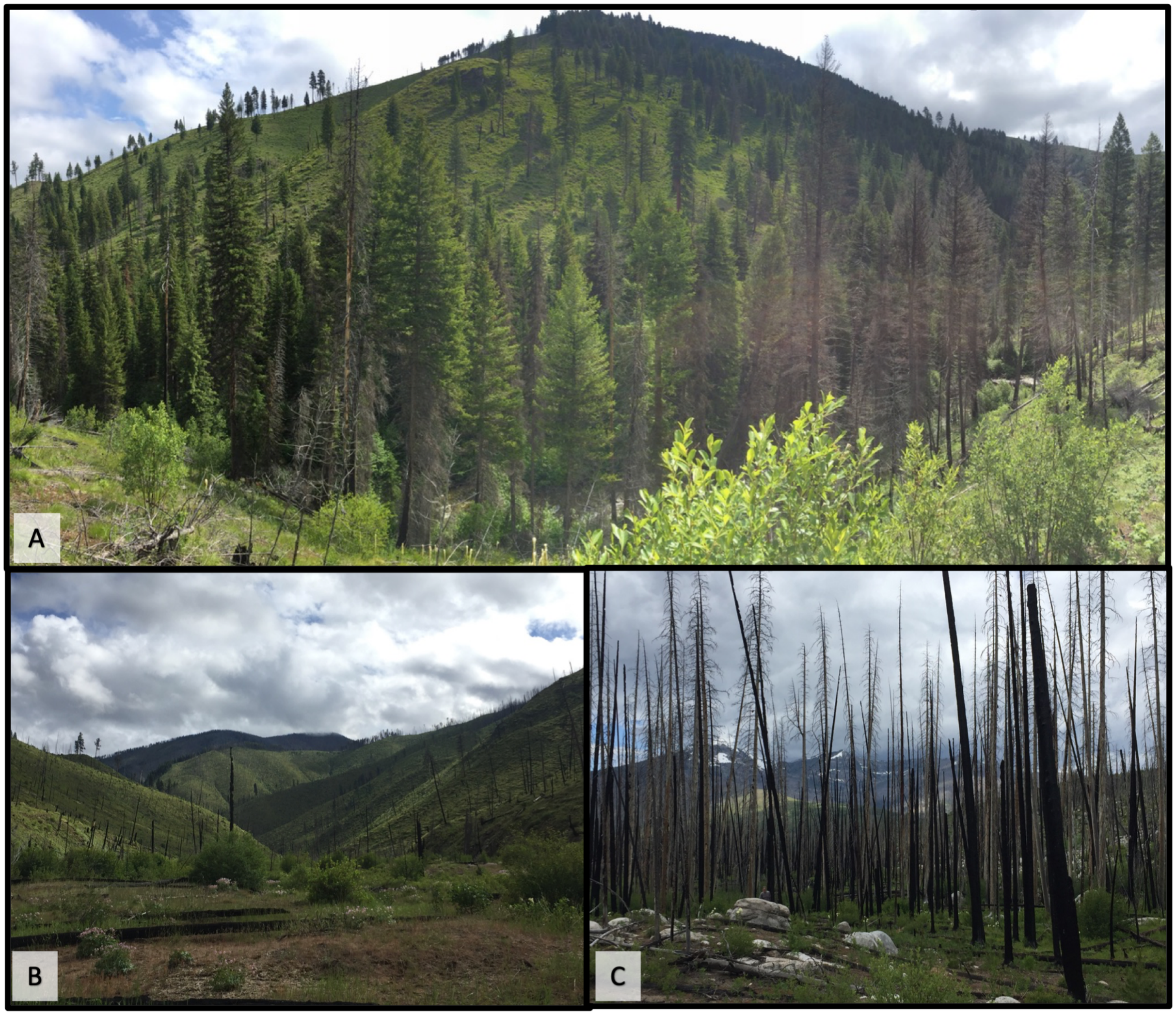

Figure 15. Stand-scale effects of fire in a) a mixed-conifer forest along the upper Selway River within the Frank Church-River of No Return Wilderness b) the low-elevation, previously forested Thompson flat area, and c) a high-elevation, recently burned site at the headwaters of the Selway River within the Frank Church-River of No Return Wilderness.

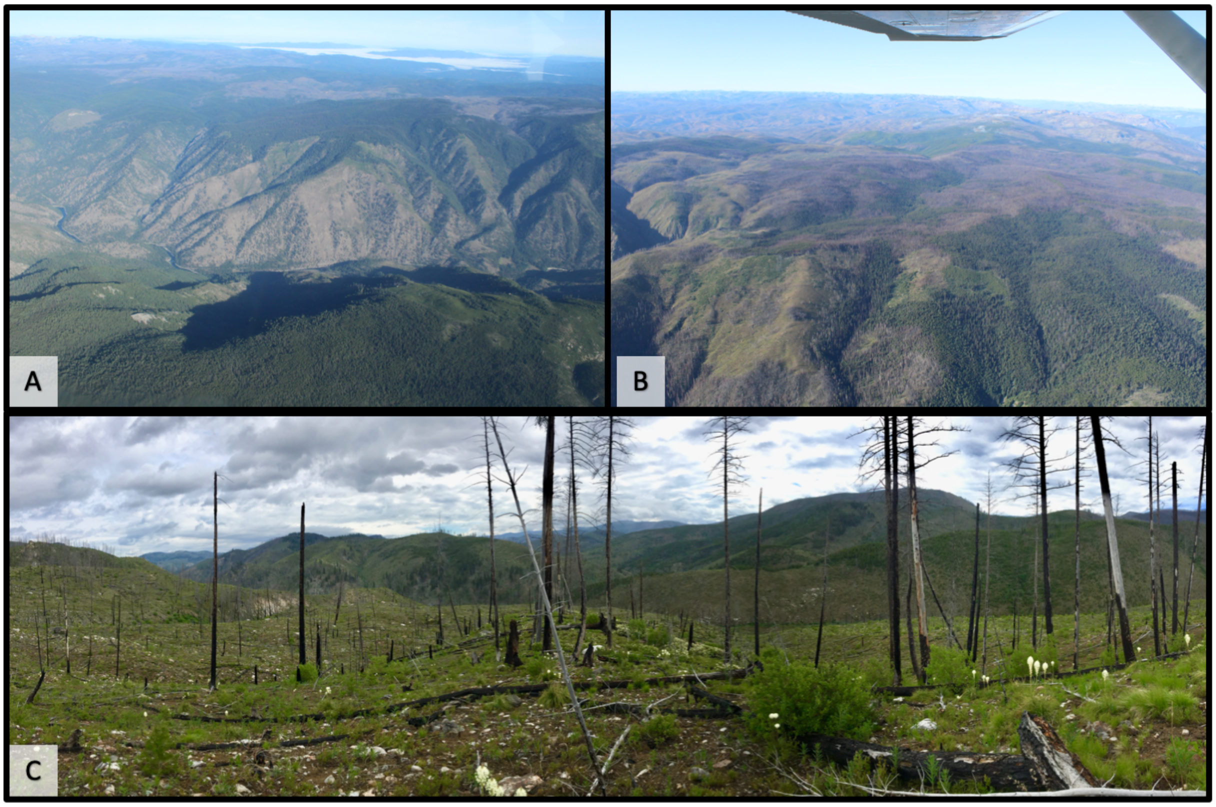

Figure 16. a & b) Aerial photography of the landscape mosaics resulting from wilderness fire in the Frank Church-River of No Return Wilderness in eastern Idaho. Photographs by K. Rahn, University of Montana. c) Landscape mosaics are also evident from the ground within the Selway River Watershed of the Frank Church. Photograph by J. Berkey, University of Montana.

Chapter 2: Applying the lessons learned from wilderness fire management will help overcome the wildland fire management paradox

ABSTRACT

Longer fire seasons and increased fire frequency associated with climate change and past suppression activities have resulted in a fire management crisis for the western U.S. Part of the solution to this crisis is wildland fire management for resource benefit, in which natural ignitions are managed for beneficial ecological effects while minimizing risk to human lives and infrastructure. To enable use of fire management for resource benefit, I draw on the knowledge gained from 35 years of fire management in three wilderness areas in the U.S. Northern Rocky Mountains. Through interviews conducted with wilderness managers with expertise in managing wildland fires for resource benefit, I found that such an approach is both feasible and desirable, but requires skilled personnel, external support, and a commitment to the ecological role of fire. I therefore recommend strengthening the workforce capacity and incentive structure for promoting fire management for resource benefit, improving lines of communication regarding wildland fire management objectives, and cultivating an ecological fire ethic within land management agencies.

INTRODUCTION

The forested landscapes of the western United States (hereafter, the West) are currently facing a fire management crisis. Record-breaking fire seasons have become increasingly common in the past two decades, resulting in devastating impacts in the form of lost lives, homes, and livelihoods (Schoennagel et al. 2017, McWethy et al. 2019). Meanwhile, continued efforts to suppress fires have resulted in the "wildfire management paradox," in which accumulated fuel loads and postponement of risk increases the likelihood of future, suppression-resistant, catastrophic wildfire (Arno and Brown 1991, Calkin et al. 2015). Confronting this challenge will require a drastic shift in how land management agencies approach fire management (Thompson et al. 2018). A number of wilderness areas and national parks in the West, however, have allowed for wildland fire management for resource benefit for decades (Pyne 1982, van Wagtendonk 2007). These areas represent a potential model for fire management that could help to solve the current fire management crisis (North et al. 2015).

In the West, fire is the dominant driver of vegetation composition and structure across most ecosystems (Wayman and North 2007, Hessburg et al. 2019). Fire also plays a key role in processes such as soil nutrient cycling (Certini 2005, DeLuca and Sala 2006, Murphy et al. 2006) and affects snow melt and accumulation through fire effects on canopy cover (Stevens 2017, Schneider et al. 2019). These fire-driven patterns and processes ultimately sustain ecosystem functions and services, including provision of critical habitat for many animal species (Hutto 1995, Smith 2000).

Despite the ecological importance of fire, management typically defaults to aggressive suppression of new ignitions (Wilson et al. 2011, USDA 2015). However, an increase in the length of the fire season and the frequency of large fires (Jolly et al. 2015, Abatzoglou and Williams 2016, Holden et al. 2018) has resulted in increased area burned in forested ecosystems of the West (Moritz et al. 2012, McKenzie and Littell 2017). This climate-driven increase in fire activity is therefore making continued suppression of all fires impossible (Schoennagel et al. 2004, Westerling et al. 2006).

Mechanical thinning, prescribed fire, and wildland fire management for resource benefit are forest management tools available for restoring the structure and function of forests while minimizing the negative effects of fire on human populations. Each of these restoration strategies, however, has limitations. For example, while mechanical thinning can be executed in a relatively controlled environment, it can be difficult to implement over large areas due to financial, legal, and operational constraints (North et al. 2014). Prescribed fire offers a less labor-intensive method for treating large areas, but lack of funds can stymie the planning or implementation processes of prescribed burns. Air quality restrictions and risk of fire escape often further limit the extent to which prescribed burning can be applied (Pollet and Omi 2002).

Managing wildland fires for resource benefit can provide a highly effective method for reducing fuel loads and restoring ecological functioning over large areas (North et al. 2012, 2015, Meyer 2014, Hessburg et al. 2015, Thompson et al. 2018). This strategy can be very cost effective, particularly when viewed as an investment to decrease future fire suppression costs (Snider et al. 2006, Houtman et al. 2013). Under the current default to fire suppression, most area burns during periods of prolonged drought, high winds, or high temperatures, when fire suppression is impossible (Reinhardt et al. 2008, Williams 2013). Managing wildfire during more moderate weather conditions, however, can limit the extent and economic impact of future wildfires by providing a fuel break and by influencing future fire occurrence and severity (Regos et al. 2014, Parks et al. 2015, 2018).

Managing a lightning-ignited fire for resource benefit comes with many challenges, especially where human populations and associated values and assets are at risk. These challenges have only increased with human population growth in fire-prone areas (Martinuzzi et al. 2015, Schoennagel et al. 2017) and increasing fire danger with climate change (Jolly et al. 2015, Abatzoglou and Williams 2016), combined with the legacy effects of past fire suppression on fuels and forest density (Covington and Moore 1994, Stephens and Ruth 2005). Institutional norms within the land management agencies and professions, as well as expectations held by the public, lead many fire managers to default to aggressive suppression of new ignitions (Steelman and McCaffrey 2011, Thompson et al. 2018). For these reasons, fire management for resource benefit is most prevalent in very large wilderness areas, where the vast amounts of land and relatively low abundance of values at risk make this approach to fire management politically and logistically easier.

Increased use of resource objective wildfire may be feasible for the large roadless areas of the West, where thinning and prescribed burning are operationally impossible, and wildfires pose minimal risk to human populations (North et al. 2012). Doing so, however, will require a dramatic cultural shift within land management agencies (Calkin et al. 2015, North et al. 2015). Because several wilderness areas have already experienced this cultural shift, it is worthwhile to look to them for lessons on how to improve wildfire management across the West. From the lessons learned over more than four decades of wilderness fire management, I provide recommendations for strengthening the use of wildland fire management for resource benefit.

BACKGROUND & METHODS

I focus on wilderness fire management in the Selway Bitterroot Wilderness (SBW), the Bob Marshall Wilderness Complex (BMWC), and the Frank Church-River of No Return Wilderness (FCRNRW) of the U.S. Northern Rocky Mountains (hereafter Northern Rockies; fig. 1). These three wilderness areas were chosen to evaluate wildland fire management for resource benefit because they each include over 1.3 million acres of wilderness land that have allowed for wilderness fire management for at least 35 years. As a result, the fire management personnel who have worked or are currently working in these wilderness areas represent a wealth of knowledge and experience in fire management for resource benefit.

Wilderness fire management within the Forest Service was born in the Northern Rockies in a remote portion of the SBW in 1972. This drainage, known as the White Cap study area, was the first Forest Service exception to the 10 AM policy that required suppression of all new ignitions by 10:00 AM the following day (Silcox 1935, Smith 2014). Following the White Cap study, the entire SBW adopted a fire management plan allowing for wilderness fire in 1978, the same year that the federal government abandoned the 10 AM policy (van Wagtendonk 2007, Smith 2014). The BMWC of northern Montana and the FCRNRW in central Idaho allowed for wilderness fire management in a portion of their wilderness area in 1978 and 1979, respectively, and by 1985 all three wilderness areas in their entirety were managing for fire. Following the adoption of these wilderness fire plans, these areas have been highly successful in returning fire to the landscape (fig. 2).

Since the 1978 revocation of the 10 AM policy, federal wildland fire policy has evolved dramatically to allow for flexible management of every wildfire for multiple objectives (USDA and USDI 2009). To understand how this evolution manifested in the successful implementation of wildland fire management for resource benefit, from June 2018 to April 2019 I conducted 20 interviews with key wilderness fire managers, both retired and current, from the Northern Rockies. The individuals selected for these interviews held a number of positions related to wilderness fire, ranging from district and forest fire management officers, district rangers, forest supervisors, regional and national fire officers, and fire research scientists. Taken together, the careers of these interview subjects spanned from the beginning of the Northern Rockies wilderness fire program in 1972 to present day.

The questions asked during the interview process were aimed at clarifying facts about the history of wilderness fire in the Northern Rockies, gaining the interviewee's perspective on the lessons learned, determining their professional incentives for allowing more fires to burn, and discussing the subject's professional opinions on how to strengthen the practice of wildland fire management for resource benefit. All interviews were open-ended and evolving, with questions uniquely adapted to the interview subject and their responses. A single interviewer conducted all interviews, which were recorded for later reference.

I present from the interview data the four primary lessons learned from wilderness fire that made non-suppression management possible and acceptable over time. I also provide the interviewees' suggestions for how to build upon these lessons to further strengthen wildland fire management for resource benefit. Finally, from these lessons and suggestions I construct a framework that provides broad, actionable recommendations for increased use of wildland fire management for resource benefit within land management agencies.

RESULTS

1. Wildland fire management for resource benefit is possible and helps achieve land management goals

Prior to reintroduction of fire in the White Cap drainage, the idea of "allowing" a fire to burn, even in a remote wilderness area, was heretical. According to one manager involved in the White Cap study, "…by 1970 we were still in the total fire suppression mode; it permeated the organization [and by] coming up with the concept [of wilderness fire] and trying to implement it on the ground, we were bucking 50 years of culture." The early wilderness blazes therefore provided evidence that fire management for ecological restoration was more appropriate than suppression in wilderness areas where values at risk and impacts on human populations were low.

This lesson that fire is a fundamental ecological process that supports other management goals has changed the way both land management agencies and the public think about fire. As one interview subject pointed out, in the era of climate change and megafires, more fire management agencies would "like to have, in and out of wilderness, more landscapes as resilient as the Frank, the Bob, and the Selway; we are paying a dear price for fire exclusion." The policy changes related to wildland fire management during the past two decades reflect this shift in thinking on the role of fire as an ecosystem process, as policy has become increasingly flexible and ecologically informed.

Despite these changes in fire policy, all interview subjects agreed that many challenges to wildland fire management for resource benefit still exist. For example, funding for post-fire restoration is often limited, especially for trail, campsite, and fence repairs. Without this funding, support for fire management for resource benefit often falters as both land managers and the public come to view fire as having a negative effect on public lands and access. Furthermore, suppression is still the default management option across the U.S. One interview subject expressed concern that land management agencies were currently moving backwards, stating that "instead of the fire program taking on more of the wilderness fire flavor, wilderness fire took on more of a suppression flavor" in the past decade. Interview subjects suggested that changing this culture of fire suppression would rely on developing incentives for wildland fire management for resource benefit, since "it is hard to find the incentive to take the risk" under the current fire policy framework.

2. Wildland fire management for resource benefit requires long-term planning and skilled personnel

After the revocation of the 10 AM policy, fire management became more nuanced. According to one early wilderness fire manager, "long duration fires that occur over 30, 60, and 90 days are so complex because the variables are so constantly changing that I've often said that it's more difficult to predict what's going to happen than it is for us to make a scientific prediction on how to go to space." Those managing these fires gained a new skill set uniquely adapted to the challenges of long-term fire management. For example, wilderness managers first had to learn to decide which fires required suppression. Although fire management plans largely dictated this decision, the process evolved greatly over time as wilderness managers learned to balance considerations for values at risk against fire-fighting resource availability.

The importance of long-term planning when making these fire management decisions quickly became evident. To circumvent resource scarcity during wildland fires being managed for resource benefit, fire personnel would often apply elements of a fire management plan far in advance. One manager described how, in 2003 in the BMWC, managers were acting “weeks ahead of time and staging equipment back there by stock … we would be doing a lot of structure protection where the cabins were wrapped for two months.”

Over time, it became apparent that the best incident management teams to implement these long-term plans were not large, suppression-oriented teams. One interviewer described the downside of suppression teams: “If they have the equipment and resources, they want to go out and touch the fire instead of just standing back and waiting and preparing.” Instead, the districts and national forests managing wilderness fire began calling in smaller teams specifically trained in long-term management strategies to execute the fire management plans. Oftentimes, these teams were pulled together for a specific wilderness fire event and would include retired management personnel well versed in long-term management strategies.

Despite the heavy reliance on those trained in long-term fire management within wilderness fire, there was a sense amongst interviewees that the skill set required for long-term fire management is on the decline within most land management agencies. One interview subject, for example, said that “we need to continually challenge our fire folks to gain those skills and get those qualifications,” while another indicated that there is “not enough mentorship going on.” Interview subjects therefore recommended that land management agencies place a greater emphasis on providing and incentivizing training and mentorship opportunities in long-term fire management.

3. Wildland fire management for resource benefit relies on support from agency leadership and the public

Interview subjects expressed a sense that support for long-term management of fire is eroding across agency leadership, which makes wildland fire management for resource benefit more difficult. As one interview subject stated, “it’s hard for somebody to come out and have full-throated ‘yes, we’re gonna do this’ if they feel like they’re all alone.” The difficulty can come from directives, such as the letter from the former Deputy Chief of the Forest Service, James Hubbard, that provided wildfire guidance for the 2012 fire season (USDA 2012). That letter, which promoted aggressive initial attack on all wildfires, resulted in many missed opportunities to manage fire for resource benefit that year. Beyond the immediate impact, however, the letter was interpreted by many to discourage long-term fire management; as one interview subject put it, “the Hubbard letter … took some of the momentum out of the program.” This sense of eroding support ultimately makes fire managers on the ground less likely to take the risk of long-term fire management.

Support from the public was identified as equally crucial to fire management for resource benefit within. Unlike with agency leadership, interviewees indicated that public acceptance of wildland fire management for resource benefit has generally increased since the beginning of wilderness fire management in 1972. As one interview subject put it, “the nation has become really sensitized to the fear of fire, and the [increased] understanding that having fire on the landscape is a way to avoid bigger fires in the future.” That support, however, has faced recent challenges, as long fire seasons become more frequent and the public is confronted with increasing impacts of smoke and risk of fire entry into populated areas. Therefore, interview subjects indicated that overcoming these challenges and continuing to build up public support will be crucial to improving wildfire management.

Ultimately, subjects identified good communication as essential for increasing support for wildland fire management for resource benefit. In the Northern Rockies, this entailed frequent updates on individual fire plans, potential fire impacts, and post-fire ecological benefits. In the BMWC, one interview subject described how “we really had to get good at providing up-to-date information on a routine basis to the public as well as political representatives so that they had a better understanding of what was going on. Even though they might not agree with what we were doing, they were informed.”

4. A wilderness ethic is important to wildland fire management for resource benefit in wilderness areas

Many interview subjects point to the language of the Wilderness Act of 1964 as inspiration for their commitment to wilderness fire management. This is commonly labeled as a strong ‘wilderness ethic,’ in which managers adhere to the language of the Act which states that wilderness should be “affected primarily by the forces of nature, with the imprint of [hu]man's work substantially unnoticeable” (*Wilderness Act* 1964). One interview subject, for example, said that he “read the Wilderness Act and believed it should be a place ‘untrammeled by man,’ and by putting the fires out I felt we were trammeling it.” Those most committed to this idea of minimal trammeling saw fire suppression as counter to their mandate as wilderness managers.

Crucial to the development of this wilderness ethic was an understanding of fire’s importance to other ecological processes. Another interview subject reminisced on “having a professor … that talked about ecology and fire ecology and that really helped set a foundation for me before I even saw fire.” This understanding of the ecological necessity of fire to the landscape helped managers incorporate fire into their own wilderness ethic, which ultimately makes them more willing to take the risk inherent in allowing a lightning ignition to burn.

Fostering a wilderness ethic, however, is not straightforward. One interview subject suggested that it is “a religion, where either you get it or you don’t.” Nevertheless, multiple interviewees commented that mentorships by previous wilderness fire managers, as well as a background in fire ecology, were crucial to their own understanding of how fire fit into a wilderness ethic. Therefore, as with the long-term management skills, interviewee recommendations for fostering a wilderness ethic ultimately boiled down to providing greater opportunities for training and mentorships in the ecological role of fire.

DISCUSSION

Here I expand on the lessons and suggestions from the interviews to provide detailed recommendations for increasing use of wildland fire management for resource benefit (Table 1). Some recommendations are best implemented at the level of local land management units, while others are better implemented through top-down leadership at the national level.

Local level

Recommendation 1: Increase support from non-fire personnel

This recommendation builds off the suggestion from interview subjects that a cultural shift in land management agencies will be necessary to decrease reliance on suppression tactics (Lesson 1). Over the past three decades, fire management has largely evolved so that fire and non-fire personnel are operating in separate spheres (Canton-Thompson et al. 2008). Increasing the amount of discourse and work shared between fire and non-fire personnel, however, could achieve greater integration of fire and land management (Schultz et al. 2019). Additionally, sharing of duties would increase the human-power and outreach capabilities for fire management for resource benefit. For example, providing opportunities to non-fire personnel to gain fire qualifications or training in public information could engender support for fire management for resource benefit.

Recommendation 2: Emphasize long-term management skills

Managing long-term fire events requires a unique set of skills and therefore an emphasis on specialized training and mentoring (Lesson 2; Doane et al. 2006, Schultz et al. 2019). Training for long-duration management already exists, as wildland firefighters can currently earn qualifications such as a strategic operations planner or fire behavior analyst. These qualifications, however, are not typically highly emphasized or rewarded (Seielstad 2015). Prioritizing these trainings for personnel from land management units with large, remote areas could strengthen their ability to manage a fire for resource benefit. Mentorships could also help with skill development, as expertise managing long-duration fires cannot be gained overnight or in a classroom (Kobziar et al. 2009).

Recommendation 3: Increase transparency with the public

Given the importance of communication to garnering support for wildland fire management for resource benefit (Lesson 3), our third recommendation is for individual land management units to train or hire specifically for increased transparency with the public. Support for wildland fire management for resource benefit increases with improved trust in land management agencies, and such trust is gained with open and honest communication (Lachapelle and McCool 2012, Rasch and McCaffrey 2019). This includes clearly communicating with the public about the overall goals of wildland fire management for resource benefit, as well as the strategies and tactics used during a specific fire event (McCool et al. 2006, Sturtevant and Jakes 2008, Meyer et al. 2015). In addition, communicating the ecological benefits of a wildfire after it burns can improve public support for the management approach (Toman et al. 2004). It is crucial that land managers have the ability to carry out the most up-to-date communication tactics, or that they hire personnel specifically for such marketing and outreach. With the proliferation of social media, communication today is very different than how it was a decade ago (Sutton et al. 2014, Zhang et al. 2019).

Recommendation 4: Promote an ecological fire ethic at the local land unit level

The importance of a strong ethical foundation to successful wilderness fire management points to the cultivation of an ecological fire ethic among fire and land managers as a means to further fire management for resource benefit (Lesson 4, Williamson 2007). An ecological fire ethic is a moral commitment to fire as an integral ecosystem process, developed from a scientific understanding of fire ecology and from practical experience on the ground. Fire managers with a well-developed ecological fire ethic will tend to view fire suppression as a last-resort option because removing fire degrades the ecosystems entrusted to their stewardship. Both mentorships and ecological training were identified as crucial to developing a strong wilderness ethic. Therefore, supporting meaningful basic fire ecology training of all fire personnel, and advanced training for upper-level managers, could promote greater consideration of fire management for resource benefit (Kobziar et al. 2009, Stephens et al. 2016).

National level

Recommendation 5: Provide more incentives for wildland fire management for resource benefit

Currently, there are many disincentives to managing a fire for resource benefit, largely in the form of risk to human lives and property loss, as well as the personal liability placed on individual fire managers (Canton-Thompson et al. 2008, Donovan et al. 2011, Stephens et al. 2016). Furthermore, there exist very few metrics of success in wildland fire management beyond the percentage of fires that are successfully contained with initial attack (Donovan and Brown

2007). The current incentive structure does not adequately reflect the recognition that fire can reduce future fire risk and support other land-management goals related to ecological outcomes (Lesson 1).

Possible strategies for incentivizing wildland fire management for resource benefit include tying measures of success more closely to long-term management goals, as stated in land and fire management plans (O'Connor et al. 2016, Schultz et al. 2019). Furthermore, within units that include this type of fire management in their land management plans, the amount of funding available for post-fire restoration of trails, roads, or fences could be tied directly to the extent to which fires are managed for resource benefit (Donovan and Brown 2005, 2007, Zimmerman et al. 2006). This would both remove the potential disincentives of loss of access or ecological degradation and instead replace it with a financial incentive for non-suppression actions.

Recommendation 6: Provide resources to support wildland fire management for resource benefit

The complexity of long-term fire management (Lesson 2) points to the need for specialized resources. In particular, creating more teams trained in long-term fire management, such as interagency wildland fire modules, would increase land management agencies' capacity to manage fire for resource benefit (Zimmerman et al. 2010). Stephens et al. (2016) further suggest that more of the members of these teams be hired on as full-time employees in order to take advantage of favorable burning conditions in the off season and increase agency capacity to plan for next season's fire management goals (see also Sneeuwjagt et al. 2013, Seielstad 2015). Finally, when fires being managed for resource benefit do require direct management action, managers often struggle to obtain firefighting resources due to the need for resources on suppression fires. This is an issue that requires further consideration to ensure that resources are available when conditions are appropriate for fire management for resource benefit (Seielstad 2015, Stonesifer et al. 2017).

Recommendation 7: Communicate the goals of wildland fire management for resource benefit

This recommendation aims to reverse the sense of eroding support for wildland fire management for resource benefit at the level of agency leadership (Lesson 3). The goals of fire management at the national level are currently ill-defined and, sometimes, contradictory (Schultz et al. 2019). Clarifying these goals is therefore an ongoing challenge for agency leadership. Doing so, however, would allow for clearer and more consistent communication of support and guidance to local-level personnel managing fire for resource objectives (GAO 2007). Similarly, strong outreach to the public from national leadership could further strengthen resource objective wildfire use. In particular, a targeted marketing campaign aimed at communicating the ecological and economic benefits of not suppressing fires, similar in scope to that of Smokey Bear, could significantly increase public support for the program (Toman et al. 2006).

Recommendation 8: Promote an ecological fire ethic at the agency leadership level

Leaders within land management agencies must interact with the politicians and legislatures whose default value structure often does not include the role of fire in ecosystems. These politicians therefore have a more negative opinion of fire management strategies that do not focus on suppression, generally resulting in political pressures towards continued wildfire suppression (Canton-Thompson et al. 2008, Steelman and McCaffrey 2011). An ecological fire ethic within national leadership therefore would be critically important in the face of political pressure and pushback against wildland fire management for resource benefit (Lesson 4).

Currently, national considerations of risk focus on the immediate risk of potential fire devastation, rather than the long-term risk of continued suppression (Busenberg 2004, Calkin et al. 2015). To overcome this, classes for leadership positions that integrate fire ecology concepts and highlight long-term management strategies will need to become a higher priority (Stephens and Ruth 2005). In addition to classes and training, however, frequent communication to agency leadership about fire management decisions from the local level could change how national leadership perceives resource objective wildfires (Sneeuwjagt et al. 2013) and increase their willingness to frequently and strongly communicate support for such wildfire management.

CONCLUSION

Wilderness areas with a long history of wildland fire management for resource benefit, such as those of the Northern Rockies, offer immense value to land management agencies and professionals because they provide guidance for managing wildfire more effectively into the future. In these areas, the culture and organization around fire management has already shifted to prioritize non-suppression tactics. As such, they offer lessons to strengthen management of wildland fire for resource benefit nationally.

Stronger support for such an approach to wildland fire management at the national level will affect the willingness of managers at local level to invest the personnel and resources into managing wildfires for ecological benefit; while focused investment at the local level will increase comfort at the national level in providing support (fig. 3). It will therefore take continued and concentrated investments at all levels within a land management agency to catalyze this shift in fire management culture.

There are several important limitations and caveats to the recommendations provided above. Risk to human life and property makes managing wildland fire for resource benefit impossible in many scenarios, and therefore these recommendations developed from large, remote wilderness areas cannot be applied in a blanket manner to all land (McWethy et al. 2019).

It is, however, an important tool for vast areas across the West where lack of accessibility, lack of infrastructure, or other constraints make thinning or prescribed burning logistically or financially impossible (North et al. 2012).

Every new fire presents a unique set of challenges in terms of firefighter and public safety, but also opportunities for managing fire to help achieve land-management goals, including fuel-reduction and maintaining ecological processes. The lessons learned from decades of wildland fire management for resource benefit in wilderness areas can be harnessed to restore fire-dependent ecosystems in the West.

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TABLES

Table 1. Summary table of the recommendations developed for strengthening wildland fire management for resource benefit. Recommendations are divided into those that are best implemented at the local level (i.e., by an individual Ranger District, BLM field office, or national park) and those that are best implemented by national leadership of the land management agencies. Each recommendation is paired with suggestions for implementation of the recommendation.

Recommendations	Examples of specific actions for implementation
Local level	
1) Increase support from non-fire personnel	• Provide opportunities to obtain an Incident Qualification Card (aka red card) • Provide training in public outreach and communication tactics
2) Acquire long-term management strategies	• Promote mentorships • Provide support for training courses specific to long-term fire management
3) Increase transparency with the public	• Hire for or train personnel in communication strategies • Utilize social media
4) Promote an ecological fire ethic at the local level	• Encourage mentorships • Add training courses and content that emphasize fire ecology
National level	
5) Provide incentives for wildland fire management for resource benefit	• Adjust allocation of post-fire restoration funding according to the fire management plan
6) Provide resources to better support wildland fire management for resource benefit	• Provide resources and funding for long-term fire management teams and post-fire recovery
7) Clearly communicate support for wildland fire management for resource benefit	• Express support for fire policy that includes fire as a process • Create a targeted marketing campaign
8) Promote an ecological fire ethic at the national level	• Provide training & mentorships

FIGURES

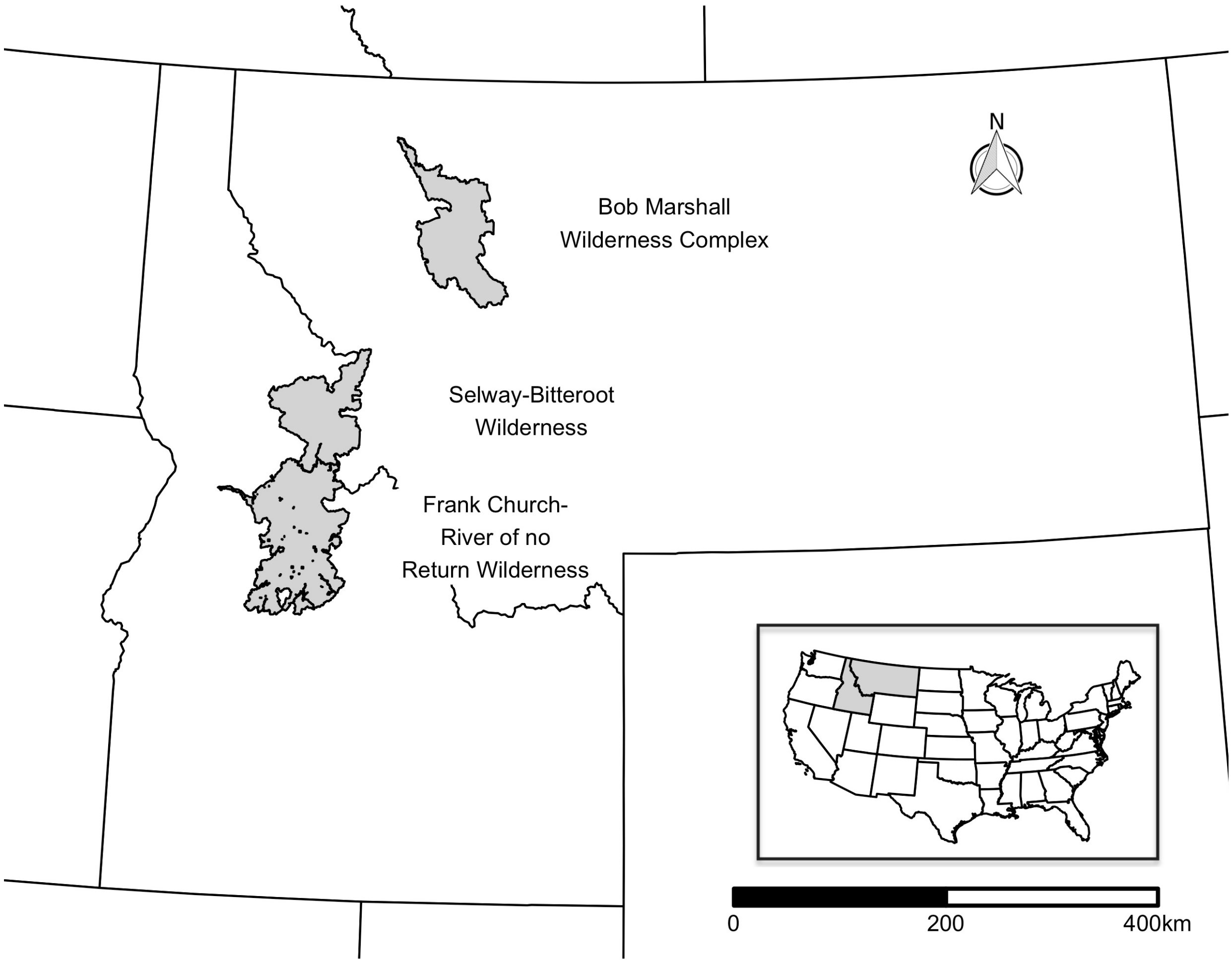

Figure 1. The three Northern Rockies wilderness areas highlighted in this paper.

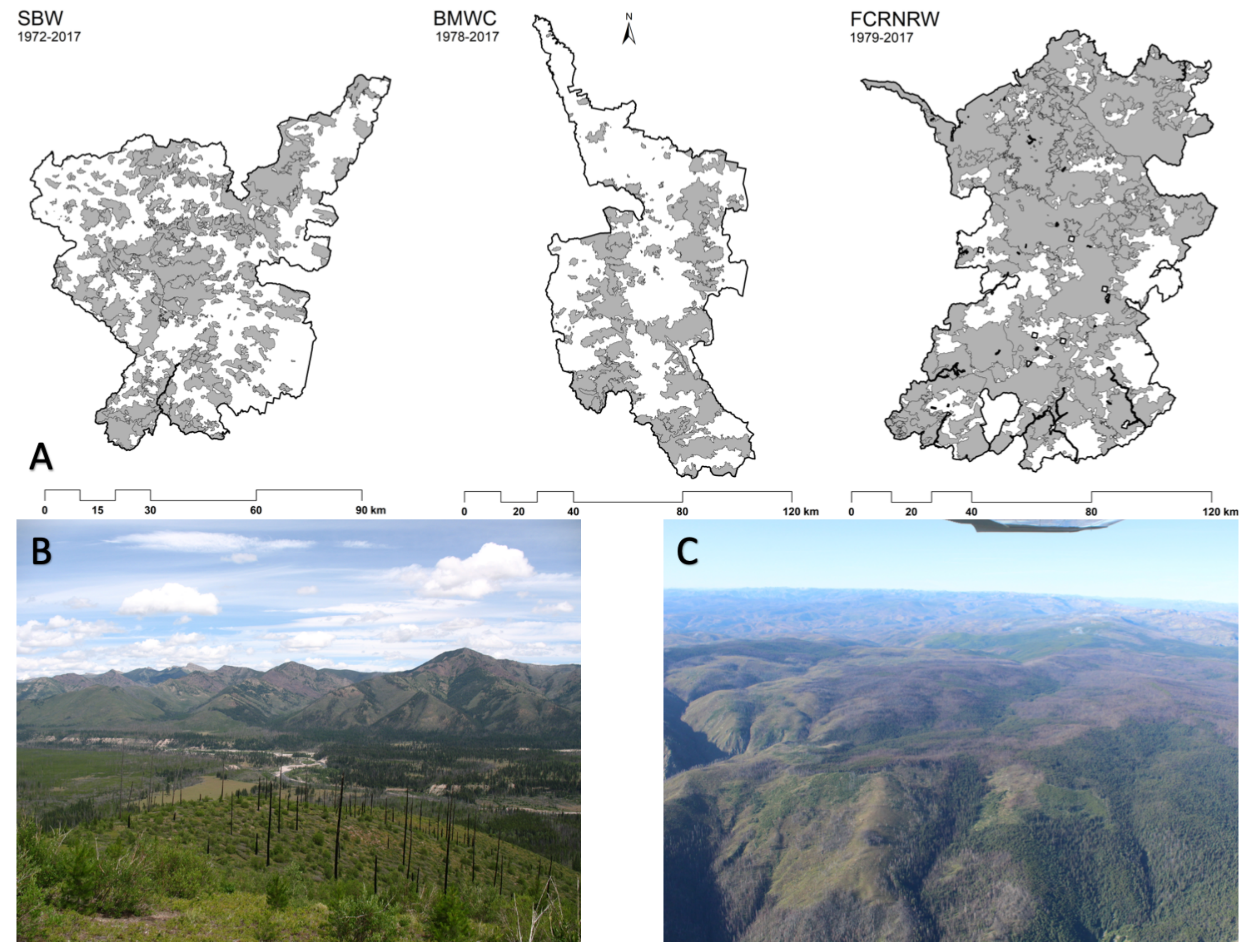

Figure 2. A) Maps of fire polygon data for the Selway Bitterroot Wilderness, Bob Marshall Wilderness Complex, and Frank Church River of No Return Wilderness following re-introduction of fire in the 1970s through 2017. B) Photo of vegetation mosaics created by fire within the South Fork Flathead River watershed of the Bob Marshall Wilderness Complex. Photo taken by J.K. Berkey. C) Aerial photo of vegetation mosaics within the Frank Church River of No Return Wilderness, captured by K. Rahn.

Figure 3. Conceptual diagram of the top down/bottom up approach to strengthening wildland fire management for resource benefit.

Chapter 3: Forest structural diversity and development in active fire regime mixed-conifer ecosystems

ABSTRACT

Nearly a century of fire suppression in the western U.S. has hindered researcher's ability to provide and rigorously test models of forest structural development in mixed-conifer ecosystems. As a result, land managers often rely on simplified, theoretical models of forest development, which ultimately hampers their ability to manage for forest resilience to future climate change. Using data from unburned (≥ 80 years since fire), once-burned and twice-burned (since 1985) mixed-conifer forests in the Bob Marshall Wilderness of Northwest Montana, I conduct a hierarchical clustering analysis to identify the forest stand structure classes resulting from 40 years of an active fire regime. I then use a suite of statistical and post-hoc analyses to elucidate the biophysical and disturbance drivers that lead to each structure class. From these results, I develop a conceptual model of forest structural development under an active fire regime in a mixed-conifer ecosystem. This model supports existing theory that succession following severe fire plays a large role in shaping forest structure, but also recognizes the role of fire at variable severities and frequencies, the physical environment, and community composition in influencing forest structural development. This conceptual model, taken with the findings of the individual analyses, highlights the complexity of forest structure and development within this system.

INTRODUCTION

Wildland fire is one of the main drivers of patch- and landscape-scale structural diversity in coniferous forest ecosystems of western North America (Habeck and Mutch 1973, Franklin et al. 2002, Hessburg et al. 2005). A large proportion of these coniferous forests are classified as mixed-conifer, or forests in which no one species dominates the species composition (Arno et al. 2000b, North and Keeton 2008). In this forest type, the range of forest structures resulting from disturbances of varying severity and frequency, as well as the developmental pathways to these representative structures, are poorly understood.

Understanding how these complex mixed-conifer forests develop is difficult due to the inherent complexity of this system. Historically, fire disturbances and post-fire successional stand development would have played a large role in shaping the structure of mixed-conifer forests across a landscape (Hessburg et al. 2015). Characterizing how fire influences stand structural development, however, is complicated by the variable fire effects experienced in the mixed-severity fire regimes that dominate most mixed-conifer forests (Arno et al. 1980, Agee 1998, Belote et al. 2015, Hessburg et al. 2016). The trajectory of structural development within a given stand is further complicated by interactions among topography, geology, micro-climate, and non-fire disturbance history (Battaglia et al. 2002, North and Keeton 2008, Lydersen and North 2012, Hessburg et al. 2015).

Previous research has attempted to explain the different forest structures present in these mixed-conifer ecosystems using theoretical models of stand development. These models tend to rely on either the paradigm of a frequent, low-severity fire regime that maintains open stands of large-diameter trees (Covington and Moore 1994, Hessburg et al. 2005), or forest successional development following a stand-replacement disturbance event (Oliver 1980, Romme 1982, Franklin et al. 2002). As a result, such models largely ignore or minimize the role of intermediate fire intensities and frequencies, as well as other environmental variables that are likely contributing to forest structure. Small sample sizes (Goslin 1997, Zenner 2005, Beaty and Taylor 2007) or a focus on only a few tree species (Weisberg 2004, Poage et al. 2009) have limited the applicability of other, field-based attempts to describe more complex pathways of forest development in mixed-conifer systems. The few studies that have attempted a rigorous, data-driven investigation of the role of fire in the structural development of mixed-conifer forests are further limited by the past century of timber harvest and aggressive fire suppression which have altered the distribution, abundance, and configuration of forest structure across large portions of the western US (Tepley et al. 2013, Reilly and Spies 2015).

Fire suppression and past land use in mixed-conifer forests has led to a profound homogenization of forest structure (Hessburg et al. 2015). Stem densities and fuel loads have increased, resulting in a decrease in vertical and horizontal complexity of forest stand structure

(Larson and Churchill 2012, Lydersen et al. 2013). The majority of this ingrowth is fire-intolerant, shade-tolerant species, which has shifted the species composition of many mixed-conifer stands (Parsons and DeBenedetti 1979, Keeling et al. 2006). At the landscape scale, the lack of fire and past logging has contributed to the loss of vegetation-type mosaics, with vast areas of land instead characterized by contiguous, even-aged forest (Arno et al. 2000b, Naficy et al. 2010, Reilly et al. 2018). Ultimately, this loss of heterogeneity has resulted in decreased resiliency of forests to future disturbances such as drought, bark beetle attack, and large, high-severity fires (Prichard et al. 2010, Lydersen et al. 2014, Hessburg et al. 2019, Restaino et al. 2019).

This decline in forest resiliency to disturbance is of increasing concern due to climate change. Across the western United States, fire seasons are predicted to last longer and burn more area in upcoming decades (Westerling et al. 2011, Abatzoglou and Williams 2016, McKenzie and Littell 2017, Littell et al. 2018). The potential impact of these shifts in fire regimes on forest structure is not well understood, but there is mounting concern about the conversion of forests to non-forest vegetation due to the combined effects of a warming and drying climate on fire activity and regeneration dynamics (Hansen and Turner 2019, Stevens-Rumann and Morgan 2019, Davis et al. 2019).

With the threat of climate change, forest resilience to wildfire has become an increasingly common goal in forest management (Rist and Moen 2013, Bone et al. 2016, Stephens et al. 2016). Achieving this goal, however, is hindered by the general lack of understanding of forest structure and structural development in mixed-conifer ecosystems within an active fire regime. An improved understanding, therefore, of how mixed-conifer forests develop in the presence of fire would allow land managers to set targets for characteristics including tree species composition and tree densities at the stand scale, and structural complexity at the landscape scale (Churchill et al. 2013, Hessburg et al. 2015). By mimicking the forest patterns and structures produced by a natural, active disturbance regime, forest managers can increase forest heterogeneity and, ultimately, resiliency as fire, insect infestations, and pathogens become less able to spread across the landscape (Rodriguez and Torres-Sorando 2001, Drever et al. 2006, Restaino et al. 2019).

The best source of information about the structure and dynamics characteristic of a natural disturbance regime for mixed-conifer forests is found in wilderness areas with a long history of managing fire for resource benefit (Collins and Stephens 2007, Rollins et al. 2011, Parks et al. 2016). Such areas are characterized by an active fire regime where fire in recent decades has played a more active ecosystem role, relative to non-wilderness areas, in terms of frequency of burning and the weather conditions under which wildfires burn (Collins et al. 2007, van Wagtendonk 2007).

With this study, I take advantage of one such active disturbance regime in the Bob Marshall Wilderness in western Montana. This area, which contains a mix of tree species exhibiting a wide range of functional traits, has allowed for management of wildland fire for resource benefit since 1981. Due to its wilderness designation, it is also free from confounding factors on forest structure and fire spread, such as road construction or vegetation management. Given these advantages, I use plot-level forest structure data from within the South Fork Flathead River watershed of the Bob Marshall Wilderness to investigate the following questions:

1. What are the potential drivers of conversion from forest to non-forest vegetation in a mixed-conifer ecosystem?
2. What forest structural classes are generated by an active fire regime?

3. What is the relative importance of fire history, topography, geology, and local climate in the developmental pathways to different forest structure classes?
4. Does live tree species composition differ between forest structure classes, and do differences in fire history explain variation in species composition within each structural class?

METHODS

Study area and site selection

All data were collected from lower slope and valley bottom sites within the South Fork Flathead River watershed of the Bob Marshall Wilderness in Northwest Montana. Elevations range from 1,266 to 1,697 m. The study area is dominated by mixed-conifer forest, with forest composition consisting primarily of lodgepole pine (*Pinus contorta*), Douglas-fir (*Pseudotsuga menziesii*), western larch (*Larix occidentalis*), Engelmann spruce (*Picea engelmannii*), and subalpine fir (*Abies lasiocarpa*, Belote et al. 2015). Ponderosa pine (*Pinus ponderosa*) is also present at low densities, primarily in low-elevation valley bottoms on well drained soils (Keane et al. 2006, Larson et al. 2013).

The study area was historically characterized by a mixed-severity fire regime, with a mean fire return interval of 20-80 years (Hopkins et al. 2014). Most fires were suppressed from 1935-1981; beginning in the 1980s, however, fire management plans changed to allow for management of naturally ignited fires for resource benefit. Since then, the area has experienced an active fire regime (Larson et al. *in press*).

A total of 224 sites were sampled between the summers of 2015 and 2017 (fig. 1). These sites fell into one of three categories: unburned (no fire since at least 1935, n = 15 plots), once-burned (one fire sometime between 1985-2013, n = 89 plots) and twice-burned (two fires between 1985-2013, n = 120 plots). Plots were selected using a stratified random sampling design with a factorial combination of number of fires, initial fire severity, and topographic classes. Number of burns and burn severity for plot selection were determined using the fire atlas published by Parks et al. (2015), updated to 2013 using data from the MTBS project (Eidenshink et al. 2007). Fire severity was grouped into four classes based on dNBR values, which were bilinearly interpolated for each plot point. Topographic classes for the plots included flat, north-east facing, and south-west facing and were bilinearly interpolated using a 30-m DEM of the study area.

Field Methods

At each plot, data were collected characterizing both mature trees and tree regeneration. To sample tree regeneration, live seedlings, which I defined as regeneration < 1.37 m tall with a diameter at breast height (dbh) of < 10 cm, were tallied by species within a 4 × 20 m plot originating at plot center. Live saplings (≥1.37 m tall with a dbh of < 10 cm) were tallied by species and sampled in a circular plot. The radius of the sapling plot depended on recent fire history, with unburned plots sampled using a 5-m radius and burned plots sampled using a 17.84-m radius.

Live and dead standing trees with a dbh of less than 80 cm but greater than 10 cm were also sampled within the 17.84-m radius plot. The species, living status, and dbh was recorded for all trees. Any trees ≥80 cm dbh within a 43.7-m radius had the same data collected to capture the density of relatively rare, large trees on the landscape.

Data reduction and analysis

Conversion to non-forest

To identify the drivers of conversion from forest to non-forest, vegetation data were reduced to include only live tree and regeneration densities for each plot. Non-forest was plots were then identified based on two definitions: 1) any plot with no live trees or regenerating trees; or 2) any plot with no live trees and ≤ 350 regeneration stems per hectare. The second definition was chosen to reflect forest management re-stocking values for the Inland Pacific Northwest and is consistent with other studies of regeneration conducted in mixed-conifer forests (Tepley et al. 2017; Povak et al. *in press*). Unburned plots were excluded from the forest-conversion analysis.

Variables included as potential drivers of conversion to non-forest fell into four categories: climatic, topographic, geologic, and fire history. Climate variables included 30-year average temperature and rainfall in the 4-km^2 pixel containing each plot, bilinearly interpolated from plot center using PRISM monthly data (PRISM climate group 2004). I also calculated the 3-year post-fire precipitation and temperature anomalies from the 30-yr average values (i.e., z-scores) for each fire at each plot. The z-scores for both the first fire and the most recent fire were included in the random forest analysis (described below).

Topographic variables were bilinearly interpolated from plot center using a 30-m DEM for the study area and included elevation and topographic wetness index (TWI). TWI serves as a proxy for shallow soil moisture levels and accounts for local controls on water movement as well as upslope water contributions (Beven and Kirkby 1979, Western et al. 1999). In addition, a heatload value for each plot was calculated using the McCune and Keon (2002) formula. Heatload is an estimate of solar radiation that incorporates plot latitude, slope, and aspect (McCune and Keon 2002).

Geologic parent material was derived, using simple extraction, from a 30-m NRCS dataset for the Bob Marshall Wilderness containing dominant landform and parent material groupings. Parent material classification included five classes: alluvial valley bottoms, cirque basins, glacial till and colluvial slopes, non-calcareous residual bedrock-controlled slopes, and calcareous residual bedrock-controlled slopes. Fire history variables included the number of burns, the fire severity of each fire, and the time since fire, calculated as the number of years between the most recent fire and the sampling date. All data extraction was performed in the program R (R core team 2017).

A random forest (RF) analysis was then conducted to identify the primary drivers of conversion from forest to non-forest. RF was chosen for this question because of its ability to handle multiple types of predictor variables and model complex non-linear relationships, including interactions among predictor variables (Breiman 2001). It is also robust to overfitting. The RF model was built using the randomForest package in R (Law and Wiener 2002). Model strength was assessed using measures of accuracy, specificity, and sensitivity.

Forest structure classes

To quantify the structure classes generated by an active fire regime, I defined forest structure as the densities of live trees, dead standing trees, and regeneration stems at each site. This definition, while not accounting for other structural elements such as species composition or downed woody debris, allows for parsimonious representation of fundamental ecological relationships between structure and function.

A hierarchical clustering analysis using Euclidean distance and Ward's linkage method (Ward 1963) was conducted on the 224 plots to identify distinct forest structure classes. The data matrix for this analysis was created by reducing all vegetation data to eight size-status variables. The size-status variables included live seedlings and saplings, as well as live and dead trees which were stratified into three dbh classes: small trees (≥10 – 30 cm), medium trees (≥30-60 cm), and large trees (≥60 cm).

The stem densities of seedlings and saplings in each plot were log transformed to better meet assumptions of normality, after adding a value of 1 to account for plots with 0 values. For tree data, the total basal area per hectare (m^2/ha) was calculated for live and dead trees, within each size class noted above. Following data reduction, the data matrix was standardized to a mean of 0 and a standard deviation of 1 (i.e., z-scores), to equalize variance among variables.

The clusters, as well as their underlying structural differences, were represented graphically with a non-metric multi-dimensional scaling (NMDS) analysis of the vegetation data matrix using the Bray-Curtis measure of dissimilarity (Bray and Curtis 1957). An NMDS was used because it allows for a visual display of distance between the clusters in multi-dimensional space in such a way that only assumes monotonicity of the underlying data (McCune and Grace 2002, Legendre and Legendre 2012). In addition, NMDS better preserves the distance between plots in ordination space than other methods. The final NMDS was conducted in three dimensions to minimize stress while maintaining interpretability (McCune and Grace 2002).

A multi-response permutation procedure (MRPP) analysis was conducted to determine if the stand structure variables were significantly different across all clusters. MRPP analysis, like the NMDS, is preferred for vegetation data because it is non-parametric test of differences between groups (McCune and Grace 2002). For this analysis, I again used the Bray-Curtis distance metric. Both the NMDS and the MRPP were conducted within the Vegan package in R (R core team 2017, Oksanen et al. 2019).

A post-hoc analysis was conducted after hierarchical clustering to investigate differences in vegetation structure across the identified forest structure classes. The data-driven delineation of classes using the clustering approach maximizes the ratio of among-cluster to within-cluster variability, and therefore invalidates formal significance testing of class-to-class differences (Huang et al. 2015). Therefore, to assess potential differences in stand structure among sites, densities were compared using descriptive statistics.

Relative importance of environmental variables to structure classes

A classification and regression tree (CART) analysis was conducted to investigate the relative importance of environmental variables in the development of the forest structure classes identified by the cluster analysis. CART was chosen because it is a non-parametric method that can handle a combination of continuous and categorical data (De'ath and Fabricius 2000). In addition, it provides an easily interpretable way to display interactions amongst variables. The CART was run as a classification analysis, in which the forest structure classes identified in the cluster analysis were the response variable.

All explanatory variables for the CART analysis were derived in the same way as the RF analysis. Fire history variables were expanded to include the number of burns since 1935 (0, 1, or 2), the severity of the first fire and second fires, the time since the most recent burn, and the time between fires. Climate variables included only 30-year average precipitation and temperature, while topographic and geologic variables were the same as was used for the RF analysis: elevation, TWI, heatload, and parent material.

I examined fire history of each forest structure class independent of the CART analysis using descriptive statistics. For each forest structure class, I investigated the proportion of plots that experienced zero, one, or two burns. In addition, I divided the time-since-burn variable into four classes, where unburned plots remained classified as unburned and all other plots were classified as short (<10 years), medium (10-20 years), or long (>20 years) time since fire.

Similarly, the fire severity variable was broken down into five classes based on field data and dNBR values: unburned plots remained classified as unburned, plots where the most recent fire resulted in a dNBR value of <200 were classified as unchanged, plots with a dNBR value between 200-400 were classified as low severity, 400-600 was classified as moderate severity, and >600 was classified as high severity.

Finally, I investigated the breakdown of the forest structure classes using a classifier that combined number of burns and burn severity. Plots were categorized as unburned, once-burned at either low or high severity, and then all possible combinations of reburns: low followed by low, low followed by high, high followed high, and high followed by low (Stevens-Rumann and Morgan 2016). Whether a fire burned at low or high severity was again determined by the dNBR value, but with the divide between low and high severity occurring at 400 (thus grouping moderate and high, from the above classification, into one class).

Vegetation composition

Further post-hoc analysis was conducted to compare live species composition and structure across the forest structure classes identified in the hierarchical cluster analysis. Because live tree structure is largely shaped by fire, the small, medium, and large live trees were further broken down into three groups based on species traits that confer fire tolerance: highly fire-tolerant ponderosa pine and western larch, moderately fire-tolerant Douglas-fir, and all other, fire-intolerant, species, following the fire resistance classification for our study area developed by Belote et al. (2015). Seedlings and saplings were also split into three groups on the basis of physiology and traits relevant to post-fire regeneration and shade tolerance: serotinous lodgepole pine, shade-tolerant species (subalpine fir, Engelmann spruce, Douglas fir, and pacific yew (*Taxus brevifolia*)), and all other, shade-intolerant species.

The same breakdown of fire history, in which plots were classified by number and order of burns at high or low severity, was again used to examine effect of fire history on live vegetation composition within the forest structural classes. In this analysis, live vegetation structure was again classified on the basis of traits that confer fire resistance for trees, and serotiny/shade tolerance for regeneration. In this case, the trees and regeneration were not divided up by size classes but were instead broken down simply into regeneration (<10 cm dbh) or trees (≥10 cm dbh). The average stems/hectare per plot for each species grouping was then compared across fire histories within each forest structure class.

RESULTS

Conversion from forest to non-forest

Conversion to non-forest was relatively rare in our dataset. Only 7% of the 224 plots had no live trees or regeneration after the most recent fire, and only 11% of plots had no live trees and less than 350 stems/ha of live regeneration. For both definitions of non-forest used in the RF models, the same five variables emerged as correlated with conversion to non-forest, although the order of these variables differed slightly: precipitation anomaly for the three years following a fire; severity of the first fire; heatload; 30-year mean temperature; and temperature anomaly for the three years following fire (fig. 2). Number of burns and time since fire did not emerge as correlated with non-forest in either analysis.

These variables cannot, however, be used to predict non-forest, as the predictive power of the RF model was low. Although the model did a fairly good job at classifying plots overall (e.g., out-of-bag error rates were relatively low), the sensitivity of the model, or its ability to predict non-forest plots correctly, was also very low for both models (table 1). For the first cutoff, where non-forest was classified as any plot with no live stems, 87.5% of non-forest plots were misclassified as forested. Ability to predict non-forest did not improve with the second cut-off, with 91.7% of non-forest plots misclassified as forested.

Forest structure classes

The hierarchical cluster analysis on the eight vegetation size-status variables resulted in six forest structure classes (fig. 3). The MRPP indicated that vegetation structure between these six forest structure classes was significantly different (A = 0.25, p value of delta = 0.001, Appendix A).

The NMDS and the descriptive statistics for the eight size-status variables used in the clustering analysis revealed the structural variables that defined each forest structure class. Forest structure class 1, for example, was identified as a unique cluster on the basis of relatively high densities of small dead trees, whereas forest structure class 2 tends to have plots with relatively low to moderate densities of all trees and regeneration (Appendix A, fig. 4). Forest structure class 3 is defined by high density tree regeneration, both in the seedling and sapling size classes (fig. 4). Plots assigned to forest structure class 4 were clustered on the basis of greater small and medium live tree densities than other forest structure classes, whereas forest structure class 6 was defined by plots with large, live trees present. The structural diversity in forest structure class 5 is high across plots (fig. 4a). Overall, however, forest structure class 5 is defined by high densities of dead trees, either in the medium or large size class (fig. 4).

Relative importance of environmental variables to structural development

The CART analysis revealed that time since fire plays a large role in determining forest structure classes (fig. 5). Plots that burned within five years tended to fall into forest structure class 2, whereas plots that have not burned since at least 1935 tended to fall into forest structure class 4. In the CART analysis, this is indicated by the branch determined by ≥ 68 years since fire.

In the plots that have burned since 1985 but have not burned within the past five years, environmental factors beyond time since fire begin to shape forest structure. These plots, which are identified by the CART tree as having burned most recently in the past 6-67 years, are classified into forest structure class 2, 3, or 5 depending on the number of burns, 30-year average temperature, and heatload. Forest structure classes 1 and 6 were not assigned a pathway to development within the CART analysis, likely because they have the fewest number of plots and therefore do not dominate any one terminal node of the CART.

The CART model reveals a number of pathways to development for each forest structure class. In particular, forest structure class 2 is identified as the predicted class in 4 of the 7 terminal nodes (fig. 5). This structure class predominated in the plots that were burned 5 years prior to sampling, or those that burned in the past 6-67 years and existed under physical conditions less favorable to vegetation growth, such as high heatload or high mean annual temperature (fig. 5). Furthermore, the overall misclassification rate for this CART model was 41.1%. This is because nearly every developmental pathway described by the CART analysis in fact contains at least four structural classes, not just the single structural class predicted by the model.

The fire history analysis revealed that many possible fire history pathways can result in a particular forest structure class. Forest structure classes 1, 2, 3, and 5 have the greatest fire activity, with only three unburned plots among these structure classes (fig. 6a). In addition, over a third of the plots in these structure classes burned at high to moderate severities at least once since 1985 (fig. 6d). Forest structure classes 4 and 6, by comparison, have a higher proportion of plots that were unburned, as well as higher proportion of plots burning only at low or unchanged severities since 1985. Across all 224 plots, only three plots burned twice at high to moderate severities, all of which fall into forest structure classes 2 or 3 (fig. 6d).

Plots in forest structure class 1 have all experienced at least one fire and tend to have burned within the past 10 years, often with at least one moderate to high severity fire (fig. 6). Plots in forest structure class 2 have also burned at least once, with the exception of one plot. This structure class also had one of the highest proportion of plots experiencing reburns (fig. 6a). These plots tended to be classified as "short time since fire" and burned at a range of severities, although over 20% of plots in structure class 2 have experienced at least one high to moderate severity burn (fig. 6d).

No plot in forest structure class 3 was unburned. Nearly a third of plots in this forest structure class have burned at high to moderate severities at least once, and these plots tend to be 10-20 years removed from the most recent fire (fig. 6). Forest structure class 5 has the greatest proportion of plots that have burned exactly once since 1985, with most of those plots burning at severities classified as low or unchanged. Only two of these plots are unburned or are classified as long time since fire.

Forest structure class 4 has the greatest proportion of unburned plots, with only three plots experiencing a high or moderate severity burn. Forest structure class 6 is similar in that it has a relatively high proportion of plots that are unburned, and seven plots that have only burned once. All structure class 6 plots that have burned did so within the past 20 years, and at severities classified as low or unchanged (fig. 6).

Vegetation composition

The post-hoc analysis of species composition and tree densities within each of the six forest structure classes revealed several distinct patterns. Forest structure classes 1, 2, and 3, for example, all have low live tree densities (fig. 7a). All non-forest plots with no live trees or regeneration (cut-off 1) fall into structure class 1 or 2. In forest structure class 3, the high regeneration densities are primarily caused by serotinous, lodgepole pine stems, although the densities of both shade-tolerant and intolerant species are relatively high in the seedling size class (fig. 7b).

Forest structure classes 4, 5, and 6, in contrast, have a much larger live tree component. In structure classes 4 and 6, the live species composition is similar for the small and medium live tree variables, with relatively low densities of fire-tolerant ponderosa pine and western larch in the small tree size class, and relatively high densities of these species and the moderately fire-tolerant Douglas-fir in the medium tree size class (fig. 7a). In forest class 6, the live, large trees are dominated by ponderosa pine and western larch, with moderately high densities of Douglas-fir and almost no fire-intolerant species. Plots in forest structure class 5 also contain moderate densities of medium and small live trees, mostly Douglas-fir (fig. 7a). Regeneration in these three forest structure classes is dominated by shade-tolerant species, especially in the seedling size class (fig. 7b).

There are some common trends across all forest structure classes in relation to plot burn history and vegetation composition (fig. 8). For live tree structure, fire-intolerant species tended to have their highest densities in plots that are unburned or burned once at unchanged to low severities (fig. 8a). The greatest densities of the highly fire-tolerant ponderosa pine and western larch species are located in plots that burned at unchanged to low severities once or twice. These fire-tolerant species only play a large role in forest structure in forest structure classes 4 and 6 (fig. 4c).

Forest structure classes 4, 5, and 6 show the greatest within-class diversity in live tree species composition (fig. 8a). In forest structure class 4, repeat burns or a single, higher severity burn generally reduced live tree densities by removing fire-intolerant species. A similar pattern is evident in forest structure class 5, although live tree stem densities were much lower overall as compared to class 4. Finally, in forest structure class 6, Douglas-fir dominated in plots that burned once at an unchanged to low severity, western larch and ponderosa pine dominated in the plots that burned twice at unchanged to low severities, and the fire-intolerant species dominated in the unburned plots.

Regeneration composition overall shows less variation in species composition between fire histories. Instead, regeneration tended to vary more between forest structure classes (fig. 8b). In structure class 3, for example, regeneration density is high across all plot burn histories. The greatest densities of serotinous, lodgepole pine regeneration in structure class 3, however, are found in plots that burned once at a moderate to high severity or in the low-high reburn plots.

DISCUSSION

Conceptual model of forest structure development

The results from these analyses point to a new conceptual model of stand development in a mixed-conifer ecosystem across a range of fire severities (fig. 9). This model is not intended to capture all of the complexity described above, but rather to describe how a range of fire disturbances, successional pathways, tree species composition, and physical environmental variables may interact to produce a mosaic of forest structure classes over space and through time.

From a successional standpoint, a high severity fire "resets" the stand to forest structure class 1 or 2. These structure classes, with their lack of live vegetation structure, correspond well with the early developmental stages following a stand-replacing disturbance event described by other forest classifications (Appendix C; O'Hara et al. 1996, Foster et al. 1998, Franklin et al. 2002, Reilly et al. 2018). Whether the stand arrives at class 1 or 2 largely depends on the pre-existing vegetation conditions, as structure class 1 requires a high density of small diameter trees prior to the fire. This structure class likely results, therefore, from a high severity burn of a relatively young, fire-intolerant stand, whereas the potential pathways to structure class 2 are more numerous.

Without further fire activity, stand development will likely lead to structure class 3. This structure class includes the "doghair" regeneration patches, typically dominated by lodgepole pine, that often result from succession and re-establishment following these stand-replacing disturbance events (Turner et al. 1997, 1999, 2004). If structure class 3 burns, it will likely reset to structure class 1, especially if that plot contained high densities of lodgepole pine that had reached reproductive maturity. If the stand burns again, or as the small, dead trees fall and becomes downed woody debris, structure class 3 could easily reset to class 2 as well. In a landscape with an active fire regime such as the Bob Marshall Wilderness, this "reverting" back to class 1 or 2 is highly likely given the predominance of fire-intolerant, lodgepole pine in structure class 3.

If forest structure class 3 remains unburned long enough it will likely develop into forest structure class 4 (Franklin et al. 2002, Turner et al. 2016). The species composition of any given stand in structure class 4, however, will vary given the species composition of the regeneration, the climate and topographic conditions, and subsequent fire activity. Fire-intolerant species, such as lodgepole pine or subalpine fir, may persist in this structure class. This persistence, however, relies on low to no fire activity and the ability to outcompete the slower growing, more fire-tolerant species (Larson 1992).

Fire activity in forest structure class 4 can lead to a number of structural changes. If it burns at high severity, the stand may return to structure class 1 or 2. A single, low severity fire in a stand may maintain the stand in structure class 4 at lower stem densities, or begin a transition towards class 6. If that fire activity is followed by a bark beetle attack, however, the trees that survived the fire may die due to beetle activity, thereby switching to structure class 5 (Ryan and Amman 1994, McCullough et al. 1998, Hood and Bentz 2007). Finally, repeat, low severity fires in structure class 4, especially if the stand has more fire tolerant species present, could cause a switch to structure class 6.

If a fire did convert a stand to structure class 6, this class is then maintained by frequent, low-severity fires. This is a well-documented disturbance history that allows for large diameter, fire resistant species such as western larch and ponderosa pine to outcompete other shade-tolerant, fire-intolerant species (Covington and Moore 1994a, Leirfallom and Keane 2010, Larson et al. 2013, Hopkins et al. 2014). A high severity fire in this class, however, could kill the large, fire-resistant trees and convert this class to structure class 5 (Larson et al. 2013). Additionally, as with structure class 4, conversion to structure class 5 could also occur with a low severity fire followed by a bark beetle attack, thereby causing mortality of large trees (McHugh and Kolb 2003).

Conversion to non-forest

Although a number of environmental variables were identified as highly correlated with the non-forest plots of this study area (fig. 2), the high error rates of the random forest models preclude conclusions on whether these environmental conditions will lead to conversion of forest stands to non-forest in the future (table 1). The low predictive power of the model may be because the dataset does not include a measure of distance to live seed source, which is often an important predictor of regeneration in non-serotinous conifer ecosystems (Davis et al. 2019). Additionally, the low number of plots classified as non-forest under both cutoff scenarios could explain some of this low predictive power. In a system where non-forest conditions are uncommon, the inherent stochasticity of regeneration makes conversion to non-forest especially hard to predict (Shibata et al. 2010).

The scarcity of non-forest plots also points to an underlying resiliency in this system. Under the climate conditions and fire activity of the past four decades, it appears that tree regeneration failure is rare, potentially because serotinous lodgepole pine is so prevalent. This species requires exposure to high heat in order to release seeds and can regenerate on landscapes impacted by large, high-severity fires (Turner 1997, Turner et al. 2004). Although experimental research suggests that there are climatic conditions under which lodgepole pine will fail to regenerate post-fire (Hansen and Turner 2019), this has not been the case in our study area over the past several decades. This mid-elevation forest type may therefore be more resilient to current climate changes than low-elevation, ponderosa pine/Douglas-fir forests (Davis et al. 2019).

Relative importance of environmental variables to structural development

The proposed model of forest development (fig. 9) relies heavily on successional theory and fire disturbances to explain development of and transitions between forest structure classes. This is in part due to the results of the CART analysis (fig. 5), where the importance of the time since fire variable in determining structure classifications highlights the relatively strong influence of post-disturbance successional dynamics in this landscape. Nevertheless, while traditional successional theory may suggest that time since fire would allow a stand to develop into a structurally diverse old forest (Franklin et al. 2002), our results suggest that these theoretical pathways of stand structure development are oversimplified for mixed-conifer ecosystems where physical environmental factors or other potential disturbances do not allow for such a predictable trajectory of forest structural development. Instead, unburned plots span forest structure classes 2, 4, 5, and 6. In fact, given the large and frequent role of fire on this landscape, none of our structural classes correspond neatly to the classification of "old forest."

Beyond the role of successional dynamics, however, the CART analysis reveals that the physical environment also plays a large role in stand structure development in stands with intermediate time-since-fire values. For example, forest structure class 3 is most likely to develop in a plot ≥ 9 years after fires (fig. 5), likely because nine years of post-fire development allows for dense regeneration to occur on suitable plots. If, however, the plot has higher annual temperatures or receives higher levels of solar radiation, then the CART model predicts it will remain in the early development stage of structure class 2. Conditions unfavorable to regeneration and tree growth maintain an early successional, open canopy stage for a longer time following the fire disturbance.

Overall, the CART model provides evidence for a system in which succession plays a large role, but structural diversity at the landscape-scale is maximized by intermediate levels of disturbance interacting with the physical environment (Roxburgh et al. 2004, Zinck et al. 2010, Hessburg et al. 2015). These results agree with previous research and theory regarding mixed-conifer systems, which states that succession and disturbance will broadly shape ecosystem structure and function while topography influences vegetation patterns at the patch scale (Hessburg et al. 2015). Furthermore, the findings underscore the complexity of this fire-driven, mixed-conifer system, as the relatively high misclassification rate indicates that stand structure cannot be reliably predicted with a few environmental or fire history variables.

Vegetation composition

There are high levels of vegetation structure and species composition diversity within each forest structure class. The post-hoc analysis of vegetation composition indicates that this diversity appears to be largely driven by fire history (fig. 8). This is important in that it further informs our understanding of within-class heterogeneity and may have implications for future forest structure trajectory. For example, the higher densities of fire-intolerant species in the forest class 6 plots that have not burned since at least 1935 suggest that these plots have experienced a buildup of ladder fuels in the understory (fig. 8). As a result, these unburned structure class 6 plots may be at greater risk of a high severity fire and subsequent conversion to a different structural class in the future (Covington and Moore 1994, Larson et al. 2013).

Similarly, there are unburned plots classified as forest structure class 2, a supposedly early development structural class. Absent of recent fire, these plots may have remained in structure class 2 rather than progressing to structure classes 3 and 4 due to conditions unfavorable for dense tree growth, such as high average temperatures or high heatloads. In this case, it is an interaction of fire history and physical environmental variables that maintains heterogeneity within a single structural class. The resulting forest stands tend to have higher densities of live trees, particularly Douglas-fir, when compared to other plots of structure class 2 (fig. 8). This is a developmental pathway not captured within our conceptual model.

Management Implications

The findings of this study underscore the complexity of managing mixed-conifer forest landscapes subject to mixed-severity fire regimes. Increasingly, land managers aim to restore heterogeneity in these systems in order to confer forest resilience to future climate change (Drever et al. 2006, North et al. 2009). What heterogenous forest structure looks like in a mixed-conifer system with fire playing an active role, as well as how these forest structures are dispersed across the landscape, is often unclear due to nearly a century of active fire suppression. The results from this study can therefore contribute to guiding management for heterogeneity and resiliency on multiple scales.

It is particularly useful to examine how the forest structural classes revealed in this analysis correspond to the structural classification guide developed by O'Hara et al. (1996). Although theoretical, the O'Hara classification system is frequently used by land managers of the Inland Northwest to guide landscape management of mixed-conifer systems (Reynolds and Hessburg 2005, Gärtner et al. 2008, O'Hara 2014), and therefore ties this research to active land management.

The comparison shows that all of the forest structure classes identified in this analysis span multiple classes in the O'Hara system, with forest structure classes 3, 4, and 5 demonstrating the greatest diversity (fig. 10). Some forest structure classes, however, do map more directly onto the existing classification system. Structure classes 1 and 2, for example, correspond well to the stand initiation stage in which a stand replacing disturbance allows for establishment of early-successional species. Forest structure class 6 similarly overlaps relatively cleanly with the old forest single-stratum class from the O'Hara model.

The lack of clean correspondence between our data-driven forest structural classes and the O'Hara classification system points to the shortcomings of relying on theoretical, simplified forest classifications to guide management. The O'Hara model attempts to categorize forest structure into discrete bins, whereas our results indicate that forest structure in a mixed-conifer forest with an active fire regime does not fit into distinct, easy-to-describe structural classes. Instead, as the structural complexity of our identified forest classes increases (e.g., structure classes 4 and 5), so too does the number of theoretical bins that that class encompasses.

The O'Hara model also downplays the role of non-stand replacing fire. Only the old forest, single stratum class of the O'Hara model explicitly recognizes nonlethal management or burning (O'Hara et al. 1996). The structural heterogeneity between and within our forest structural classes, however, relies on a range of disturbance severities and frequencies.

Finally, the overlap of multiple of our classes within the O'Hara classification system points to the relative importance that our model places on dead, standing structure. Forest structure class 1 ultimately differs from class 2 due to the high densities of small, dead trees within these plots. Similarly, structure class 5 is distinguished from structure class 4 in our analysis due to differences in large, dead stem densities. The differences in dead structure between these classes will have implications for future fire activity (Larson et al. *in press*) as well as wildlife habitat for species such as the black backed woodpecker (Hutto 1995, Hoyt and Hannon 2002).

CONCLUSION

This is the first study to capture the heterogeneity of forest structure and development within a mixed-conifer ecosystem with an active fire regime. As such, it offers new insight into the complexity of the mixed-conifer ecosystem. The frequent focus on certain archetypes of fire-forest dynamics, such as stand replacing fire or frequent low severity fire, makes this study particularly important to management of fire-driven landscapes that fall somewhere in the middle of that continuum (Agee 1993, Hessburg et al. 2016). The complexity of fire history and its impacts on forest structure within this system indicates that these extremes are not a particularly useful framework for thinking about mixed-conifer landscapes as a whole. Instead, it is necessary to manage these forests in the context of a system in which succession from a stand-replacement fire event is heavily shaped by the physical environment in combination with subsequent, non-stand replacing disturbance events, such as low severity fire or bark beetle (Hood and Bentz 2007, Lydersen and North 2012, Hessburg et al. 2015, Merschel et al. 2018).

Within this framework, the results from this study can broadly inform management of mixed-conifer ecosystems, particularly at the landscape scale. I provide an improved understanding of the structural diversity present in a mixed conifer system with an active fire regime. This includes not only the forest structure classes identified in this study and their developmental pathways, but also the heterogeneity contained within those structure classes. The range of fire severities, as well as the range of fire frequencies and the interaction with fire and the physical environment, contributes to a high level of diversity in the system. Managing for these levels of heterogeneity should increase forest resilience as competition for resources decreases and fire and disease become less likely to spread (Rodriguez and Torres-Sorando 2001, Drever et al. 2006, Restaino et al. 2019).

Ultimately, however, these findings do not offer a prescriptive restoration or management strategy. The inherent complexity of this system makes that impossible, as there is no way to predict the “appropriate” stand structure or composition on the basis of a few environmental variables. Instead, the complexity of the results highlights the importance of, when possible, restoring the natural process of fire across a range of physical conditions, as attempting to otherwise engineer the broad range of forest structural patterns will likely fall short. These findings can therefore be used to guide policy writing and decision-making regarding fire and forest management for forest resiliency to climate change.

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TABLES

Table 1. Error rates for the random forest analysis of conversion for cutoff 1 (non-forest = no live stems on the plot) and cutoff 2 (non-forest = no live trees and ≤350 regeneration stems/ha). A "0" indicates a non-forest plot and a "1" indicates a forested plot.

Cutoff 1				Cutoff 2			
Out-of-bag error rate = 7.18%				Out-of-bag error rate = 11.5%			
	0	1	Error Rate		0	1	Error Rate
0	2	14	87.5%	0	2	22	91.7%
1	1	192	0.5%	1	2	183	1.1%

FIGURES

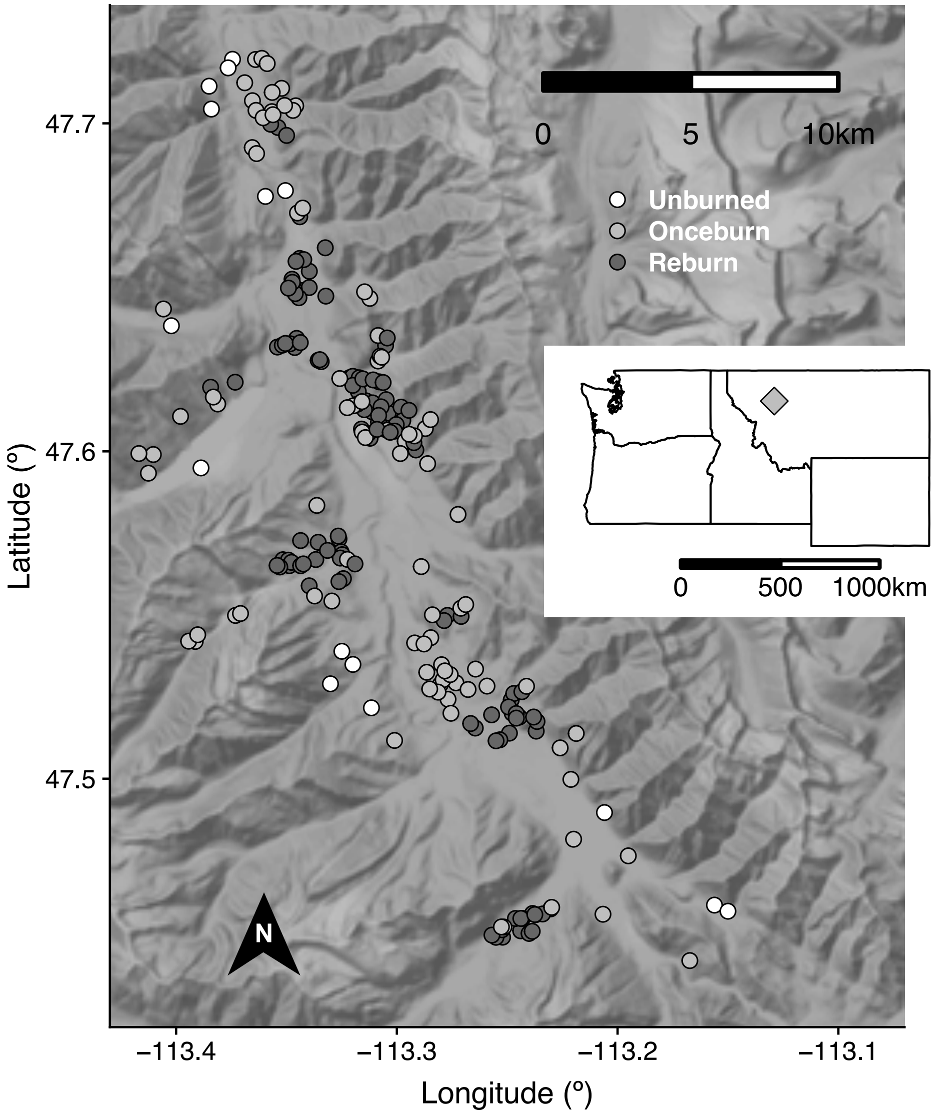


Figure 1. Plot sampling locations within the South Fork Flathead River watershed of the Bob Marshall Wilderness in northwestern Montana.

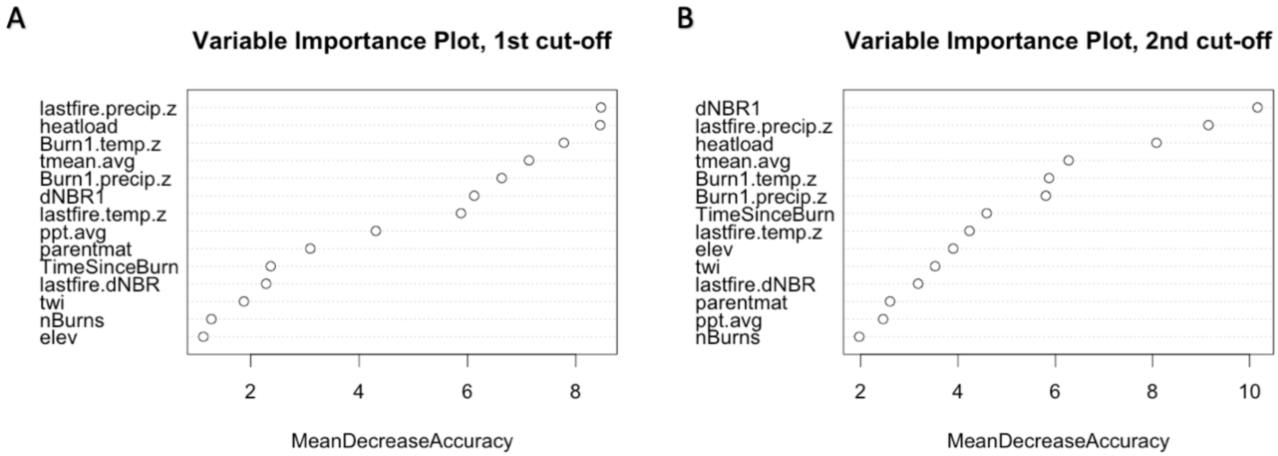


Figure 2. Ranked importance of environmental variables in predicting conversion of a plot to non-forest for a) the first cutoff, where non-forest = no live stems on the plot and b) the second cutoff, where non-forest = no live trees and ≤350 regeneration stems/ha. A higher mean decrease in accuracy score indicates that that when that variable was excluded from the random forest decision making process the ability of the model to classify a plot as forested or not decreased.

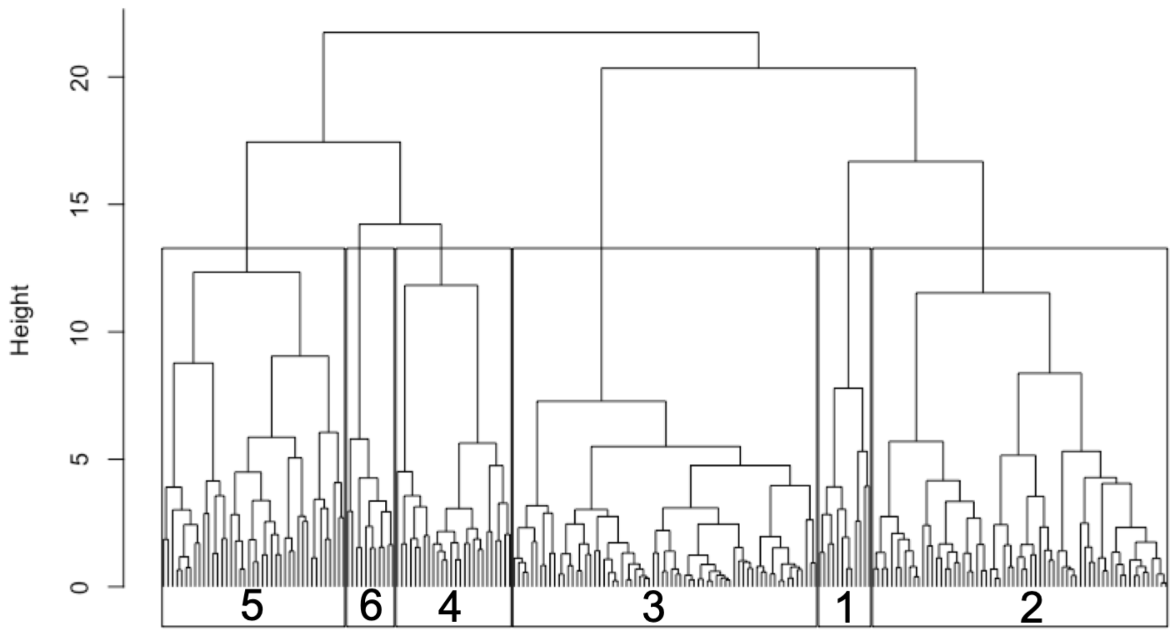


Figure 3. The dendrogram resulting from the hierarchical clustering of 224 mixed conifer plots on eight vegetation status-size variables. 12 plots were assigned to forest structure class 1, 66 to structure class 2, 68 to structure class 3, 26 to structure class 4, 41 to structure class 5, and 11 plots were classified as forest structure class 6.

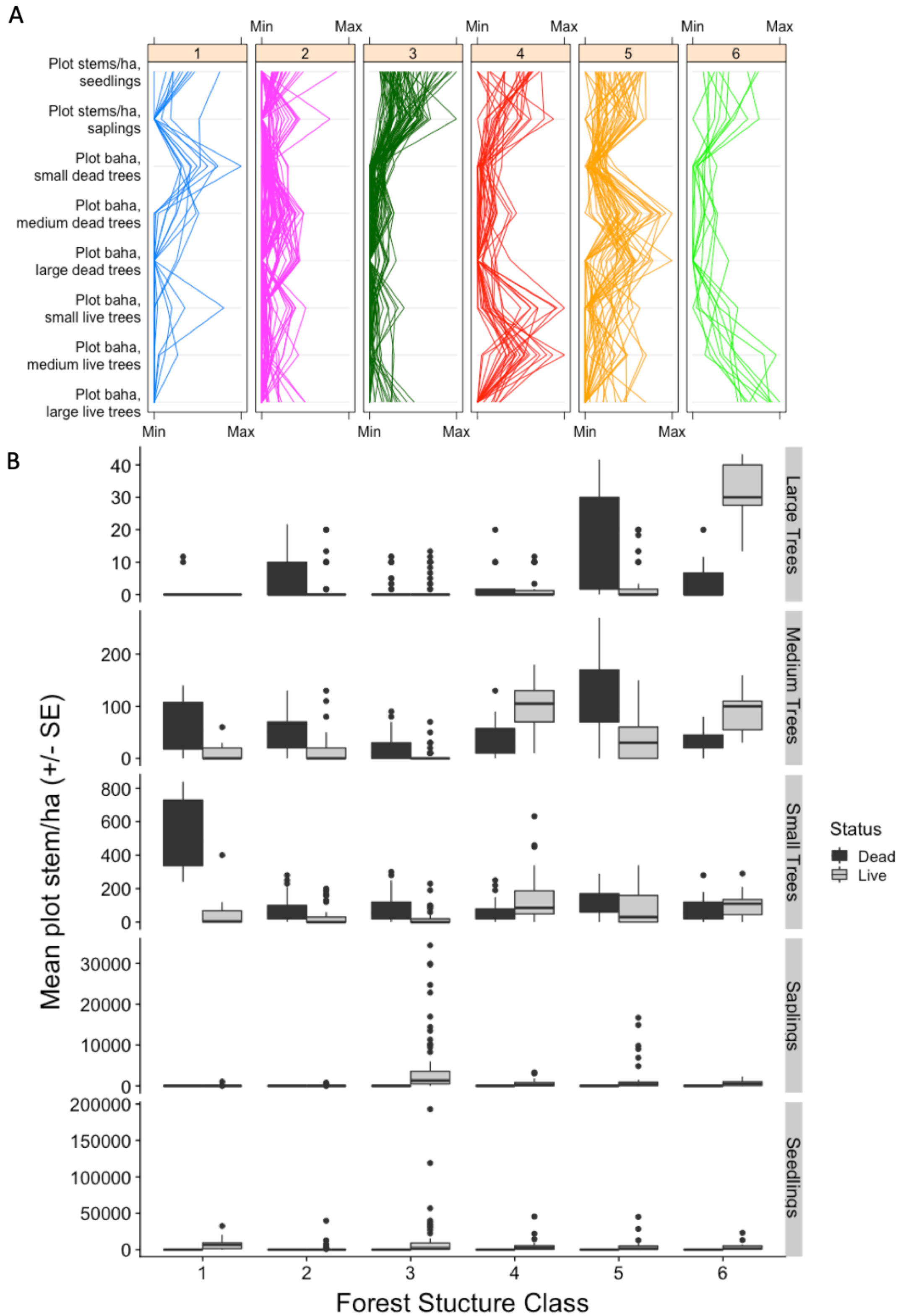


Figure 4. The vegetation structure of the six forest structure classes based on the size-status variables used for the hierarchical cluster analysis. Panel A displays structure relative to the minimum and maximum densities across all

224 plots, where each line represents the forest structure of a single plot. Panel B displays the structure in terms of the average and standard errors of raw stem densities.

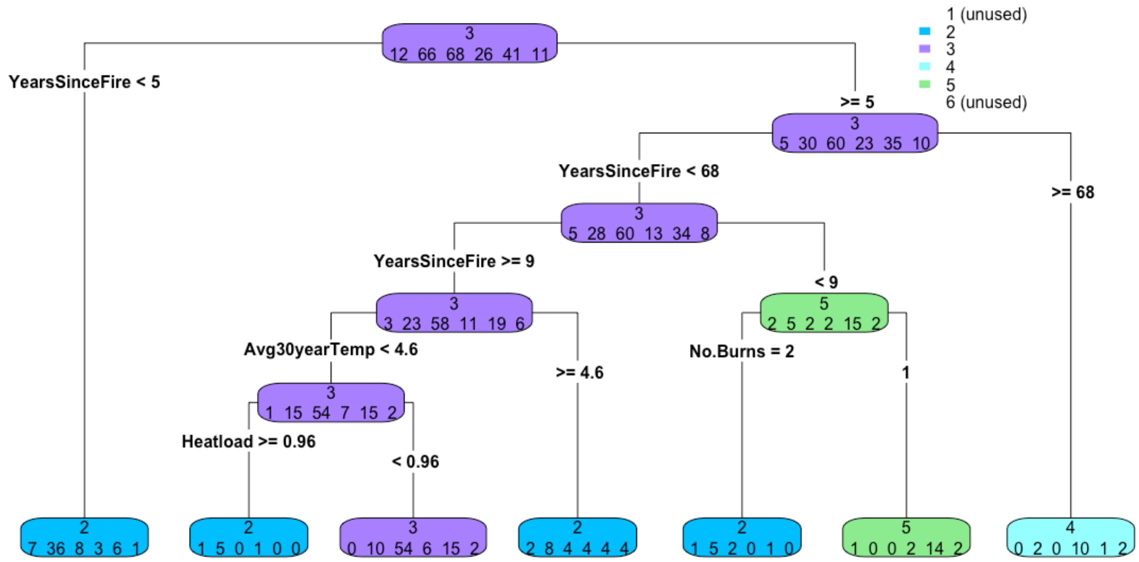


Figure 5. Results of CART model, indicating the most likely forest structure class for a given combination of environmental variables. The top number of each box represents the predicted forest structure class, while the numbers at the bottom of the box represent the number of plots within each structural class, ordered 1 through 6, that have that combination of environmental variables.

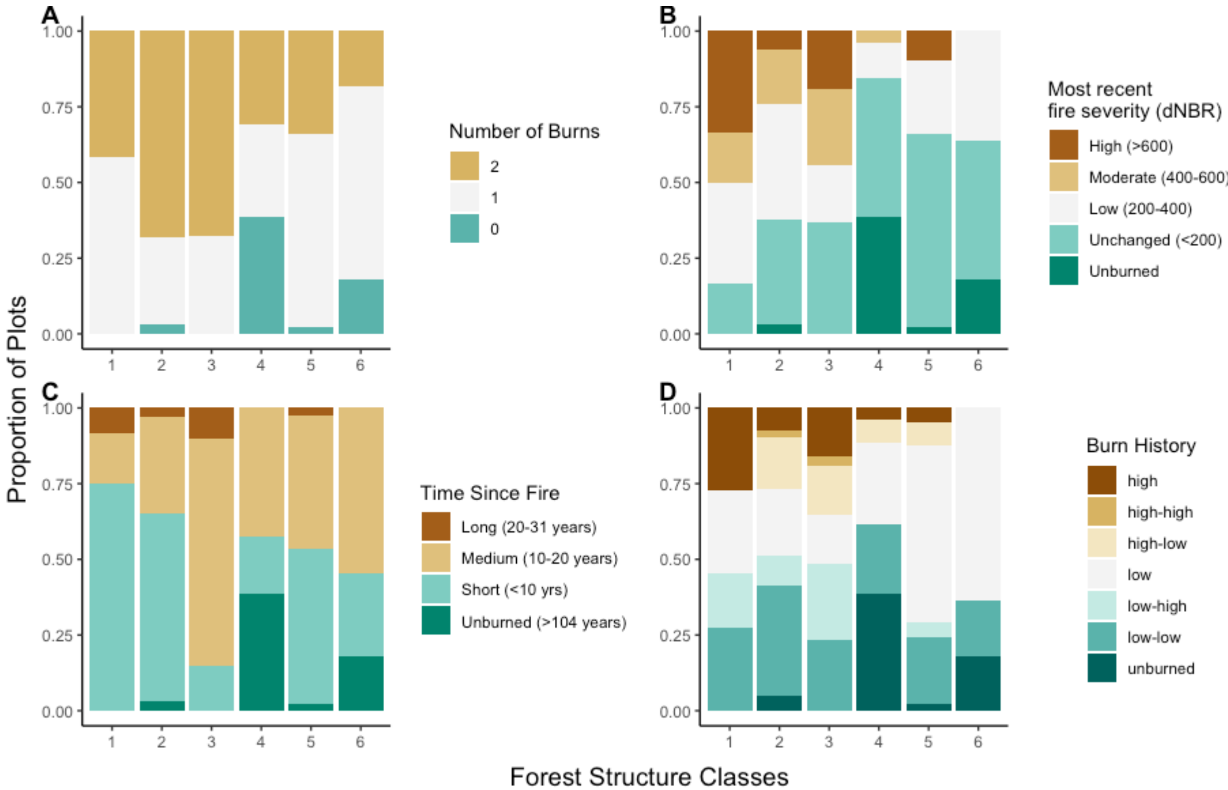


Figure 6. Fire history, including a) number of burns, b) the severity of the most recent fire, c) the time since fire, and d) the sequence of burn severities for each forest structure class. In panel D, the cut-off between low and high severity was a dNBR value of 400.

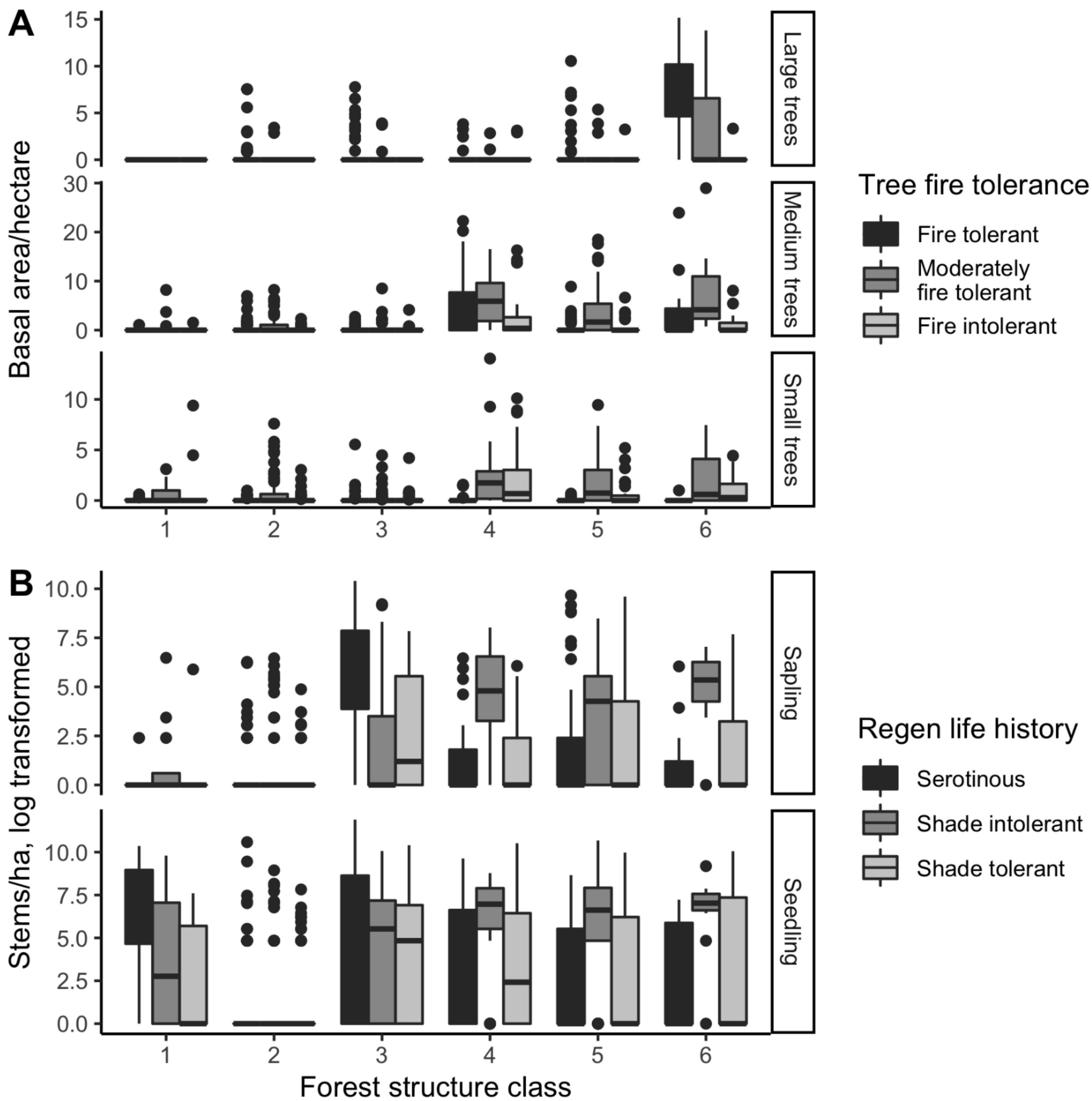


Figure 7. The live species composition and structure of the six forest structure classes identified in the hierarchical cluster analysis. Tree species were classified a fire tolerant (ponderosa pine or western larch), moderately fire tolerant (Douglas-fir), or fire intolerant (all other species). Regeneration species were classified as serotinous (lodgepole pine), shade intolerant (subalpine fir, Engelmann spruce, Douglas fir, and pacific yew), and shade intolerant (all other species). Note the differences in density measurements between A) trees and B) regeneration plots, which is reflective of the variables used for the clustering analysis. The scales also differ across the y-axes.

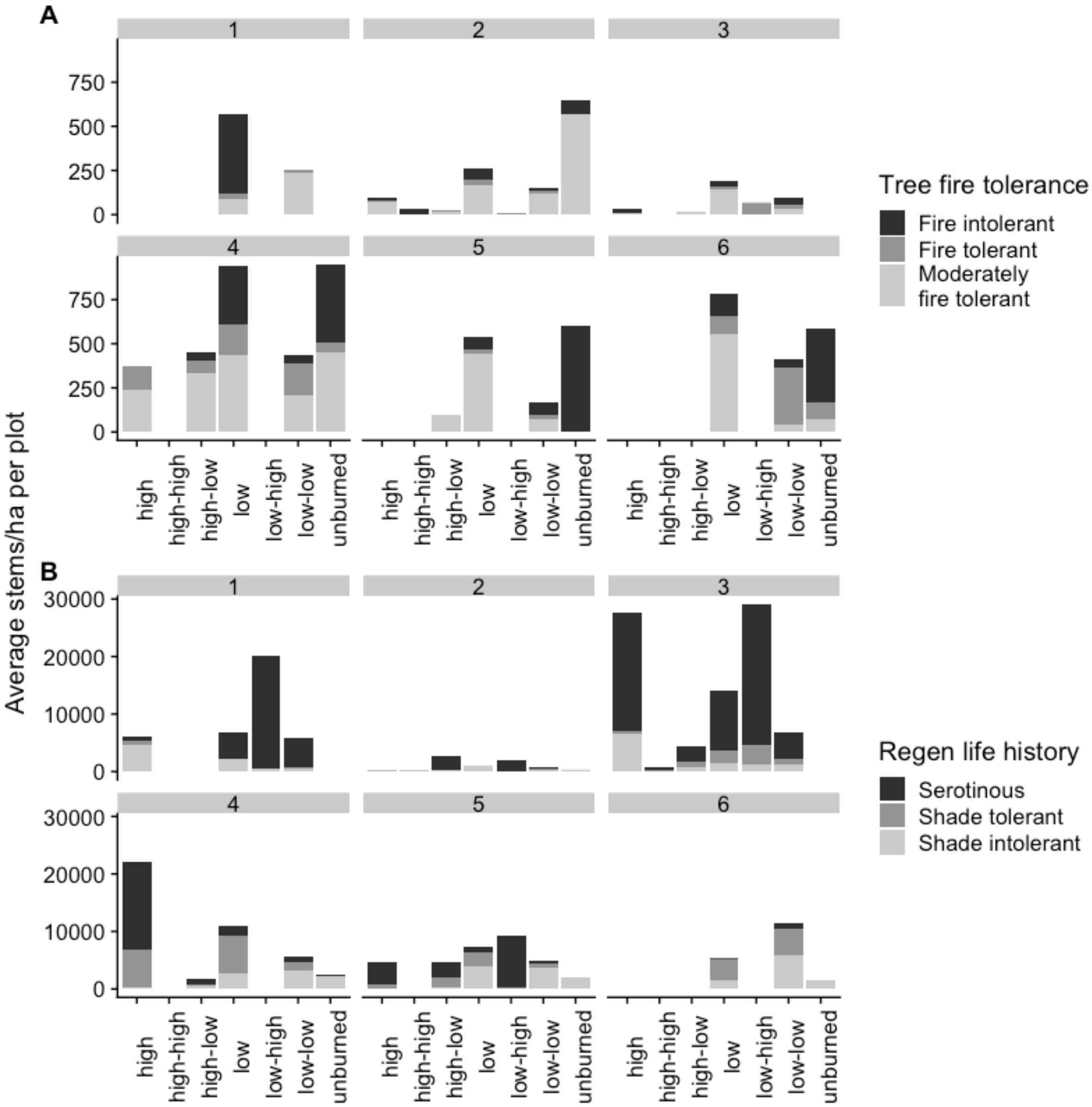


Figure 8. Live vegetation composition for A) trees and B) regeneration for each forest structure class, broken down by fire history. Fire history was determined by the dNBR value for the plot, with 400 serving as the cutoff between a low and high severity fire.

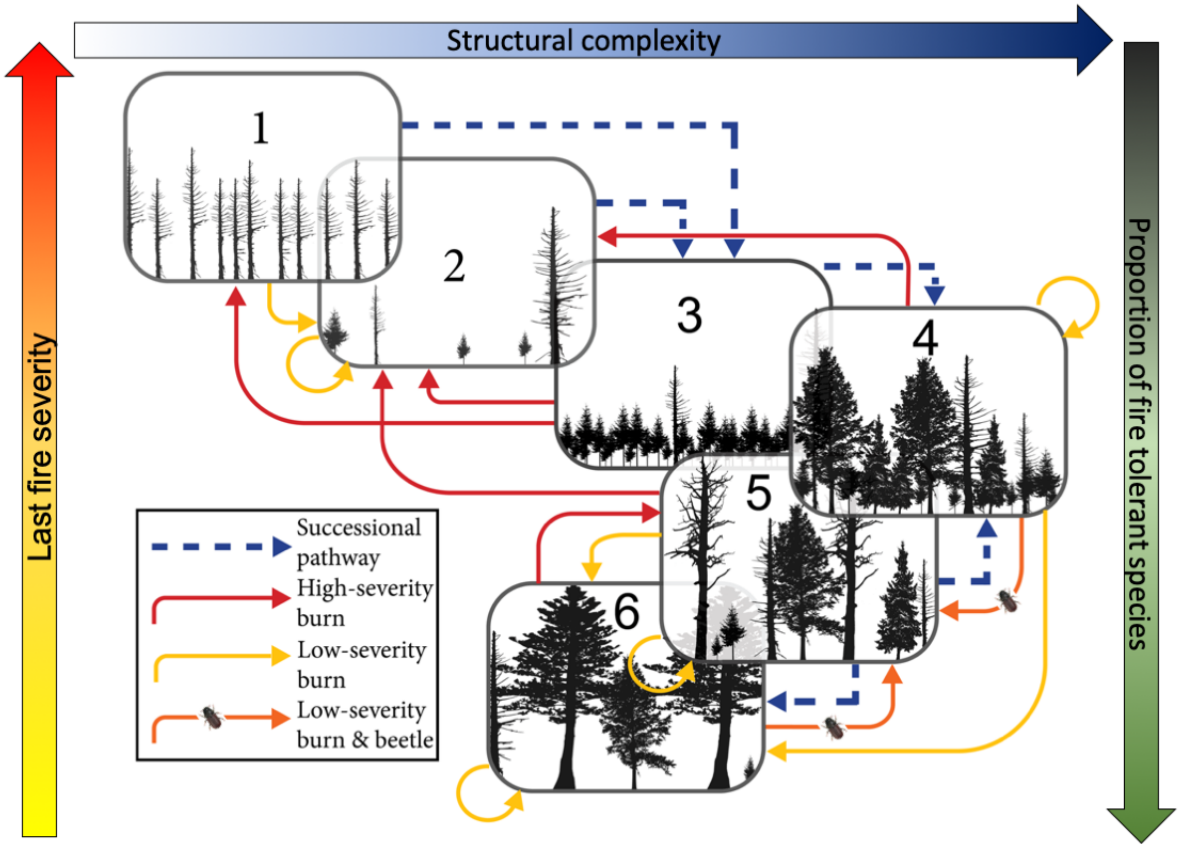


Figure 9. Conceptual model of stand development in a mixed-conifer ecosystem with a resumed fire disturbance regime. For each axis, the variable increases towards the head of arrow- i.e., the last fire severity increases going up vertically, the structural complexity increases from left to right, and the proportion of fire tolerant species increases going down vertically.

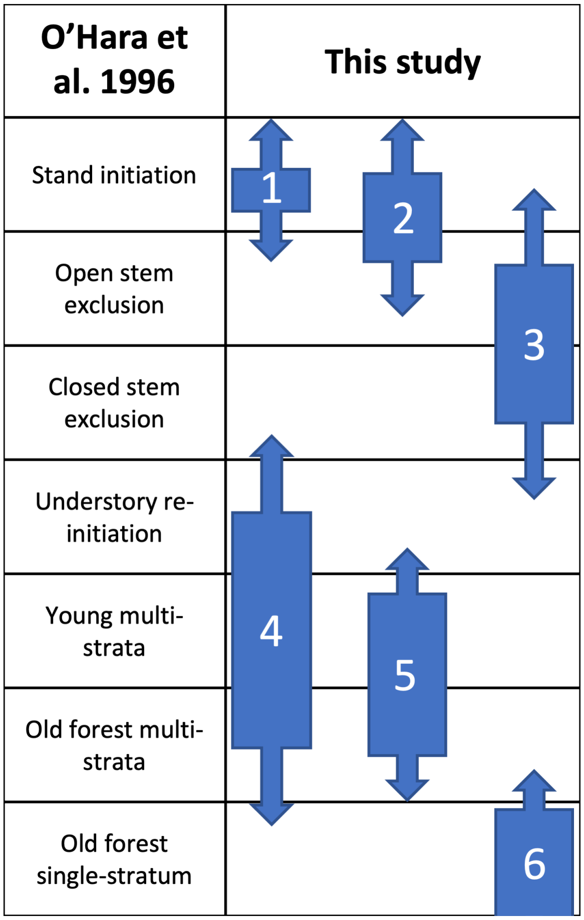
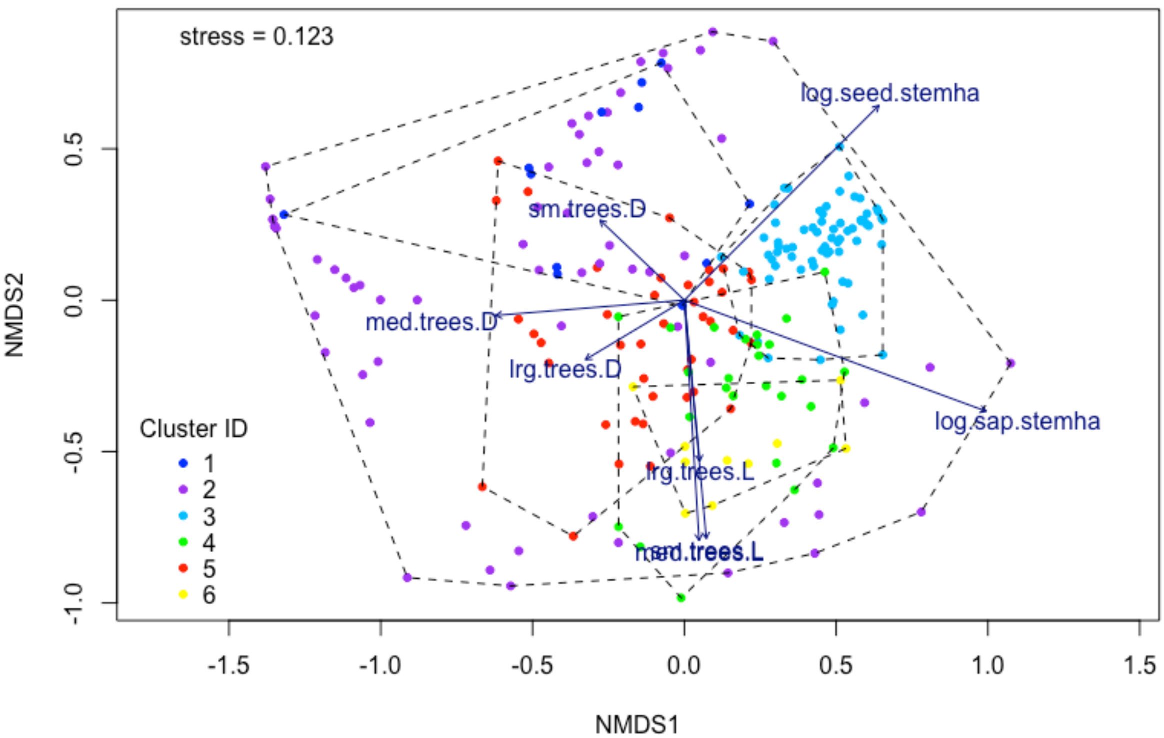


Figure 10. Comparison of our structure classes to those of O'Hara et al. (1996).

APPENDICES



Appendix A. The NMDS for the 224 plots, conducted on the vegetation data matrix of eight vegetation size-status variables in three dimensions. The final NDMS solution was was centered and rotated orthogonally to its principal components (Legendre and Legendre 2012), resulting in a display of the data in two dimensions. In this plot, the first axis contains the highest dispersion of points and the second axis contains the next greatest dispersion (Oksanen et al. 2015). Arrows depict the magnitude and direction of the correlation between each size-status variable and the ordination axes.

Appendix B. Summary table of mean and standard deviation of vegetation densities for each size-status variable used in the clustering analysis, by forest structure class. Vegetation density was measured using basal area (BA) per hectare for trees and stems per hectare for regeneration.

Cluster	No. plots	Tree Size Class	BA (live, m^2/ha)	Std. dev.	BA (dead, m^2/ha)	Std. dev.	Regen Size Class	Stem/ha	Std. dev.
1	12	Large	0.0	0.0	0.8	1.8	Sapling	88	293
		Medium	1.3	2.5	8.3	7.2	Seedling	8,469	9,797
		Small	2.0	3.6	15.9	6.8			
2	66	Large	0.6	1.6	1.8	2.6	Sapling	62	160
		Medium	1.5	2.6	6.7	5.1	Seedling	1,274	5,204
		Small	1.0	1.8	2.5	2.0			
3	68	Large	0.8	1.8	0.5	1.3	Sapling	4523	7776
		Medium	0.6	1.6	2.4	3.1	Seedling	11,869	28,707
		Small	0.6	1.2	2.3	2.2			
4	26	Large	0.8	1.4	0.9	1.7	Sapling	627	903
		Medium	13.8	7.0	5.1	4.7	Seedling	5,538	9,595
		Small	4.9	4.6	2.4	2.2			
5	41	Large	1.4	2.7	6.1	4.8	Sapling	1,851	3,938
		Medium	5.1	5.9	16.8	9.8	Seedling	4,585	8,307
		Small	2.4	2.7	3.5	2.5			
6	11	Large	11.4	2.5	1.5	2.4	Sapling	662	706
		Medium	13.6	7.6	5.0	3.8	Seedling	5034	7,167
		Small	3.3	2.4	2.6	2.7			

Appendix C. Plot level photographic representations of each forest structural class, 1-6, taken in the field.