

Fire History (1889–2017) in the South Fork Flathead River Watershed within the Bob Marshall Wilderness (Montana), Including Effects of Single and Repeat Wildfires on Forest Structure and Fuels

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Abstract—Wilderness areas offer value to society as a source of scientific information. We used fire perimeter records from the upper South Fork Flathead River watershed (Montana) to characterize the area burned one or more times during three periods: the pre-fire exclusion period (1889–1934), the fire exclusion period (1935–1980), and the fire management period (1981–2017). We also quantified the effects of a recent reburn on forest structure and fuels using a before-after-control-impact study design. Total area burned and area burned multiple times depended strongly on time period. The active fire regime during the fire management period mirrored total area burned and area reburned in the pre-exclusion period. At once-burned sites, fuel loads for most fuel types increased or were stable from 2011 to 2015, reflecting ongoing deposition of fire-killed branches and trees. In contrast, the second fire either reduced or maintained surface fuels in 2015 relative to 2011 levels. Seedlings decreased significantly in the twice-burned plots while there was no change in once-burned plots; live overstory tree densities were stable over time in both once- and twice-burned plots. Managers can use the results presented here to inform the design and monitoring of forest landscape restoration prescriptions.

Keywords: fire effects; fire management; reburns; wilderness management; wildland fire use

INTRODUCTION

Wilderness areas offer value to society as a source of scientific information. Large wilderness areas provide unparalleled opportunities to develop and test scientific theories about the causes and consequences of natural disturbances (Miller and Aplet 2016). For example, they have been critical for testing theory about self-limiting wildfire severity and spread (Collins et al. 2007; Parks et al. 2015b), river channel dynamics and forest-stream interactions (Hauer et al. 1999; Montgomery and Abbe 2006), and couplings

between upland wildfires and fluvial habitat dynamics through the delivery of sediment and large wood to the channel network by debris flows (Benda and Bigelow 2014). Much of the scientific value of large wilderness areas is derived from the untrammelled character of disturbance-driven landscape systems, for example, active fire regimes in which lightning-ignited wildfires are allowed to burn, and large alluvial rivers with unimpeded flow and channel migration (fig. 1). Intensively managed lands in which fires are suppressed, hillsides are logged, rivers are dammed,

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Figure 1—Aerial oblique view (looking south) showing the 2013 Damnation Creek fire reburning an area of old-growth western larch/mixed-conifer forest previously burned by the 2000 Helen Creek fire alongside the South Fork Flathead River, Bob Marshall Wilderness, Montana, USA. Much of the scientific value of large wilderness areas is derived from the untrammled character of disturbance-driven landscape systems; for example, active fire regimes in which lightning-ignited wildfires are allowed to burn, and large alluvial rivers with unimpeded flow regimes and channel migration (photo: J. Flint, USFS).

and geomorphic processes are altered by road building do not provide the same scientific opportunities as large wilderness areas for understanding natural disturbance processes.

The use of wilderness areas as scientific observatories is consistent with the purpose and intent of the Wilderness Act of 1964. Yet wilderness areas have long been underutilized for scientific purposes (Franklin 1987), even as their potential scientific value to society grows due to land conversion, increasing human population, and anthropogenic climate change. For example, wilderness areas provide control areas with which to compare active management strategies for climate change adaptation (Belote et al. 2015a,

2017). Research in wilderness also contributes to the design and improvement of sustainable forest management practices used outside wilderness areas, including silvicultural treatments that produce commercial timber (Hopkins et al. 2014).

Since the early 1980s (fig. 2), managers have allowed many lightning-ignited fires to burn with minimal interference in forests of the Bob Marshall Wilderness (BMW) in northwestern Montana (Smith 1986). This accumulated mosaic of fires affords important opportunities to investigate wildfire effects on forest structure, postfire tree regeneration, and fuel loads in forest ecosystems.

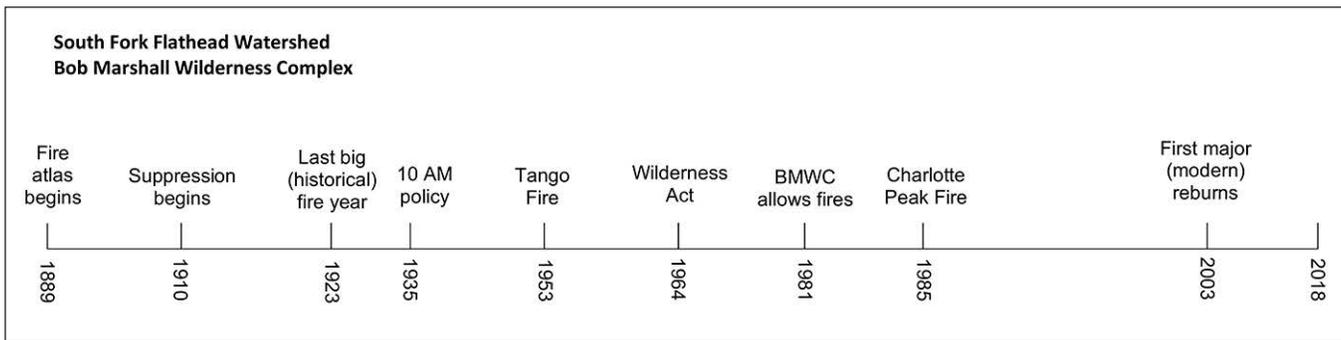


Figure 2—Timeline of fire and fire management activity in the upper South Fork Flathead River watershed within the Bob Marshall Wilderness, Montana, USA. Key management changes include implementation of the 10 AM policy in 1935, and the decision to allow for fire management within the Bob Marshall Wilderness in 1981.

Of particular scientific interest are the comparative effects of single and repeat wildfires (also called reburns) on forest vegetation and fuels. Short-interval reburns (less than about 25 years between fires in western conifer forests) can have strong effects on fuels (Stevens-Rumman and Morgan 2016; Ward et al. 2017), vegetation composition and structure (Coop et al. 2016; Coppelatta et al. 2016), and postfire successional trajectory (Larson et al. 2013). One challenge to studying reburn effects is the inability to impose experimental control over wildfire events. Consequently, most reburn studies have been retrospective, with no experimental control or prefire measurements. This is particularly true in wilderness areas, where regulations prohibit experimental manipulations.

The subject of this study is the fire history since 1889 in the upper South Fork (SF) Flathead River watershed within the BMW, including effects of single and repeat wildfires on tree regeneration, forest structure, and fuels. Our first objective was to characterize the area burned one or more times in each of three management periods: the pre-fire exclusion period (1889–1934), the fire exclusion period (1935–1980), and the fire management period (1981–2017). Our second objective was to investigate the effects of a recent reburn event on tree regeneration, forest structure, and fuels using a before-after-control-impact (BACI) (Green 1979) study design in old-growth western larch (*Larix occidentalis*)/mixed-conifer forest.

METHODS

Study Area

The study area comprises the portion of the upper SF Flathead River watershed (above Bunker Creek) within the BMW, Montana, an area of 222,243 ha. Elevation along the main stem of the SF Flathead River within this area ranges from 1,183 to 1,436 m; maximum elevation within the watershed is 2,834 m. Forest composition within the valley is dominated by lodgepole pine (*Pinus contorta*), Douglas-fir (*Pseudotsuga menziesii*), western larch, Engelmann spruce (*Picea engelmannii*), and subalpine fir (*Abies lasiocarpa*), with minor amounts of ponderosa pine (*Pinus ponderosa*) (Arno et al. 2000; Belote et al. 2015b; Keane et al. 2006; Larson et al. 2013). High elevation sites support whitebark pine (*Pinus albicaulis*) and alpine larch (*Larix lyallii*), and fires in the BMW burn to the alpine treeline (Cansler et al. 2016), where they influence structural complexity of the alpine treeline ecotone (Cansler et al. 2018). Native Americans used the area more or less continuously from at least 1665 to 1938 based on tree ring dating of bark peeling scars on old ponderosa pine trees (Östlund et al. 2005).

Fire History

We divided the fire management history for the study area into three time periods: pre-exclusion, exclusion, and fire management. We defined the pre-exclusion period as all years before 1935. Although the creation of the Forest Service, Department of Agriculture in 1905 meant that some backcountry fire suppression activity did occur between 1905 and 1934, it was

largely ineffective due to lack of personnel and technology (Koch 1935; Pyne 1982). Instead, climate largely drove fire activity prior to 1935 (Heyerdahl et al. 2008; Morgan et al. 2008).

The exclusion era began in 1935 (fig. 2) with the initiation of the 10 AM policy at the national level, which stated that all fires should be attacked with the purpose of suppression before 10 o'clock the following morning (Silcox 1935). The increase in firefighting crews and equipment during this period, combined with a climatic shift to generally cooler springs and wetter summers, made backcountry fire suppression more effective (Morgan et al. 2008; Pyne 1982). However, following the passage of the Wilderness Act in 1964 and the subsequent success of the White Cap wilderness fire management program in the Selway-Bitterroot Wilderness (Idaho and Montana), the 10 AM policy was abandoned in 1978 (Smith 2014; van Wagendonk 2007; Wilderness Act 1964). By 1981, managers began allowing some lightning-ignited fires within the BMW to burn in the Danaher Creek drainage, thus beginning the fire management period in our study area (fig. 2). A fire plan for the entire BMW was put in place in 1983 (Flathead National Forest 1983a,b).

We compiled preexisting fire atlases for the northern Rocky Mountains (Gibson et al. 2014; Parks et al. 2015a). The Gibson atlas provided fire perimeters for 1889 through 1978 for the study area, whereas the Parks atlas covered 1979 through 2012. We then updated the fire perimeter data to include fires through 2017 using BMW fire perimeters obtained from Spotted Bear Ranger District fire management staff. All fire perimeters were clipped to the upper SF Flathead River watershed boundary (upstream of Bunker Creek), then clipped again to the BMW boundary, and divided by management period for analysis. This allowed us to determine the spatial and temporal differences in area burned one or more times during different management periods.

Field Methods

All fuels and forest structure measurements were made in $n = 20$ plots, half of which were located in the vicinity of Little Salmon Park on the west side of the SF Flathead River (once-burned plots), and the

other half of which were located in the area around the confluence of Damnation Creek and the SF Flathead River on the east side of the river (twice-burned plots). Plot locations were randomly distributed along an approximately 3 km reach of the main valley, centered on the coordinates of 47.66165°N, -113.34091°W and ranging in elevation from 1,340 m to 1,600 m. In the area sampled by our field plots, the west side of SF Flathead River burned in the 2003 Little Salmon Complex Fire. The east side of the river burned in the 2000 Helen Creek Fire, and again in the 2013 Damnation Fire (fig. 1). The area west of the river did not burn a second time. All three fires were ignited by lightning. Multiple cohorts of 200- to over 700-year-old western larch dominated the overstory of these mixed-conifer forests, with lodgepole pine, Douglas-fir, subalpine fir, and Engelmann spruce making up the rest of the tree community. These study areas were selected in an earlier study of postfire tree mortality in old-growth western larch/mixed-conifer forests (Belote et al. 2015b).

We used spatial partitioning of fire events and repeated measurements of plots to establish our BACI design. In 2011, 10 plots were established and sampled on each side of the river to characterize the severity and effects of the 2000 and 2003 fires (Belote et al. 2015b). Half of these plots reburned in the 2013 fire (fig. 1). In 2015, all plots were relocated using global positioning system coordinates and remeasured to compare the twice-burned area on the east side of the corridor to the once-burned area on the west side of the corridor. We used the before reburn (2011) and after reburn (2015) measurements as our before and after with the once-burned plots as our control and the twice-burned plots as the impact.

We censused seedlings, saplings, and live and standing dead trees for all tree species within each plot. For seedlings (<1.37 m tall), we recorded the height class (0–40 cm, 40–80 cm, or 80–137 cm) and species of stems within four 1-m-radius subplots which were centered 6 m north, east, south, and west of plot center, as well as within a 1-m-radius subplot at plot center. To inventory saplings (>1.37 m tall and <20 cm diameter at breast height [d.b.h.]), we recorded the diameter class (0–5 cm, 5–10 cm, or 10–20 cm), status (alive or dead), and species of all saplings within 17.84

m of plot center. For overstory trees (stems ≥ 20 cm d.b.h.), we recorded the species, diameter, tree type (live standing tree, dead standing tree, or uprooted or snapped (or both) below d.b.h. but inferred to have been standing at time of fire) within 17.84 m of plot center. Additionally, we recorded trees with a d.b.h. greater than 80 cm within 43.7 m of plot center.

To inventory fine wood debris (FWD), we recorded fuels transects based on the planar intersect technique of Brown and Van Wagner (Brown 1974; Van Wagner 1968, 1982). Each plot had four transects which ran north, east, south, and west from plot center. Along each transect, we counted the number of intersections of 1 hour (0–0.64 cm) and 10 hour (0.65–2.54 cm) fuel particles from 3 m to 6 m from plot center. Likewise, we counted the number of intersections of 100 hour fuels (2.55–7.62 cm) from 3 m to 9 m from plot center. We also measured litter (undecomposed organic material) and duff (partially decomposed organic material) depths at 3 m and 9 m from plot center along each transect.

To inventory coarse woody debris (CWD; >7.6 cm diameter), we measured the large-end diameter, small-end diameter, and length of all woody debris particles within the perimeter of a 6-m-radius subplot with its origin located at plot center. If a piece of woody debris tapered to a diameter less than 7.6 cm, the small-end diameter and length were measured only up to the point at which the debris still had a diameter equal to or greater than 7.6 cm. If a piece of woody debris extended beyond the boundary of the 6-m-radius subplot, we recorded only the length within the boundaries of the subplot. We recorded species (if identifiable) and decay class (1–5, with 1 indicating a sound log with no decay and 5 indicating a very decayed log).

Field Data Analysis

We summarized fine fuel (1–100 hour) loads for each plot using Brown's (1974) equations for mixed-species fuels. We classified all CWD as 1,000 hour fuels. To estimate 1,000 hour fuel loads, we approximated the volume of logs as a conical frustum, and estimated wood densities by decay class using values for conifer wood from Liu et al. (2006). Because Liu et al. (2006) used four decay classes, we used the density value from their fourth decay class for our classes 4 and 5.

We tested for significant differences in four BACI contrasts using permutation tests where we randomly shuffled before reburn/after reburn and once-burned/twice-burned labels among plots 10,000 times (Roff 2006). Our contrasts were differences in means ($n = 10$ plots) of response variables between twice-burned after reburn and twice-burned before reburn (Impact After – Impact Before; IA – IB), once-burned after reburn and once-burned before reburn (Control After – Control Before; CA – CB), before reburn twice-burned and before reburn once-burned (Before Impact – Before Control; BI – BC), and after reburn twice-burned and after reburn once-burned (After Impact – After Control; AI – AC). We calculated two-tailed P-values as the ratio of the number of values at least as large in magnitude (absolute values) as observed values to the number of simulations (10,000). We repeated these analyses for seedling, sapling, and tree (live and dead) densities, fuel loads in each fuel size class (1–1,000 hour), and litter and duff depths. All analyses were performed in the R environment (R Core Team 2018).

RESULTS

Fire History

Our assessment of area burned from the fire history maps revealed that 127,327 ha (314,632 acres) burned from 1889 through 1934 (pre-exclusion), only 585 ha (1,446 ac) burned during the exclusion period, and 117,489 ha (290,321 ac) have burned since the beginning of the fire management period (fig. 3). Our analysis identified 1889, 1910, 2003, and 2017 as major fire years for this study area, or years when the area burned exceeded the 90th percentile of annual area burned from 1889 through 2017 (fig. 4). During these 4 years, 150,709 ha (372,410 ac) burned, which constitutes approximately 61 percent of all area burned over the course of the study period.

The total area and annual rate of area that reburned in the fire management period (9.3 percent cumulatively) were similar to the amounts in the pre-exclusion period (7.6 percent), although there was less area burned three or four times during the fire management period than in the pre-exclusion period (table 1). In contrast, only 0.3 percent of the total area burned, with no reburns, during the exclusion period (fig. 5, table 1). Fire rotation was 79 years, 17,096 years, and

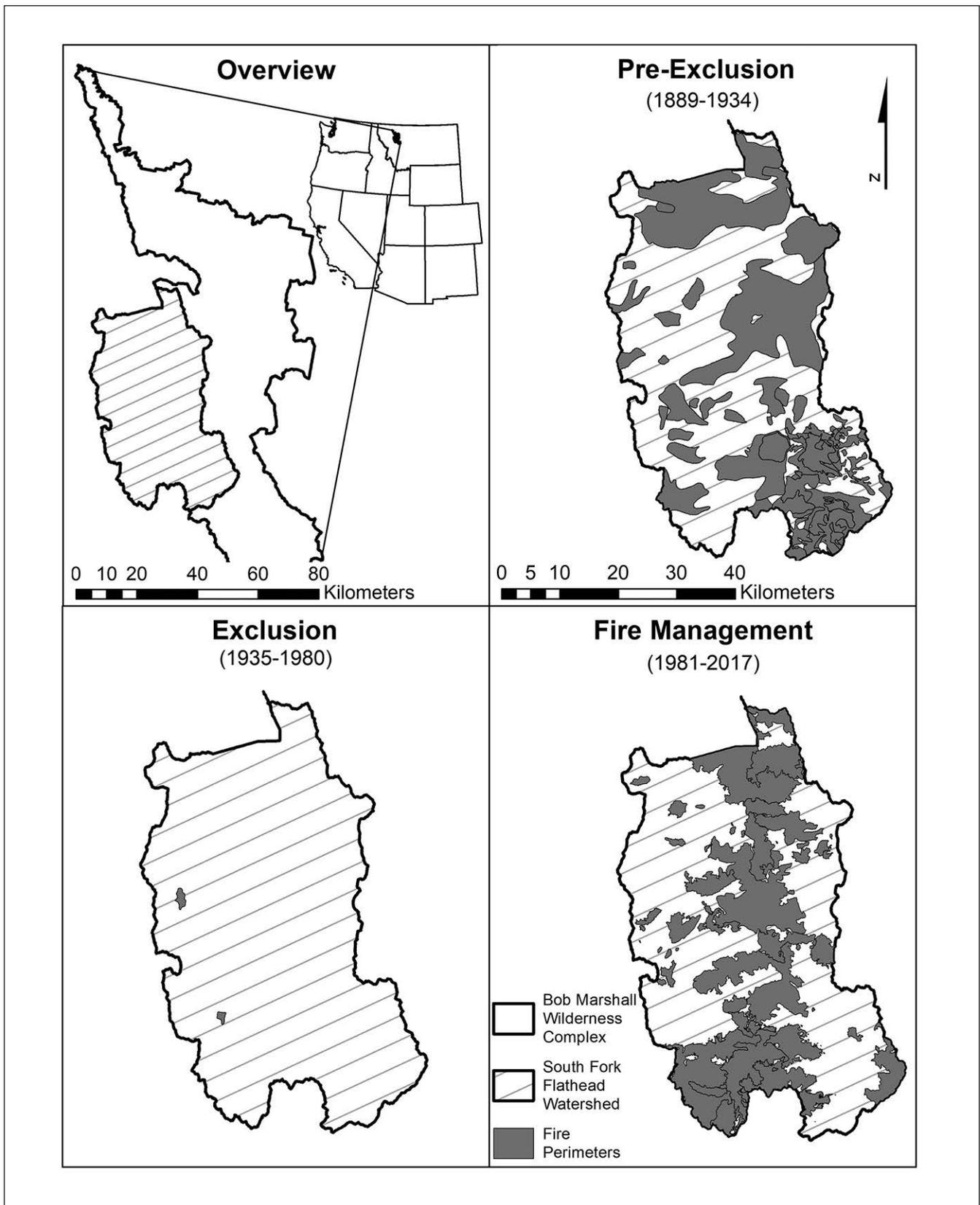


Figure 3—Maps of area burned within the upper South Fork Flathead River watershed of the Bob Marshall Wilderness, Montana, USA during the three contiguous management periods: pre-exclusion (1889-1934), exclusion (1935-1980), and fire management (1981-2017). Fire extent is greatly reduced during the fire exclusion period due to a combination of climatic shifts and increased backcountry suppression activity.

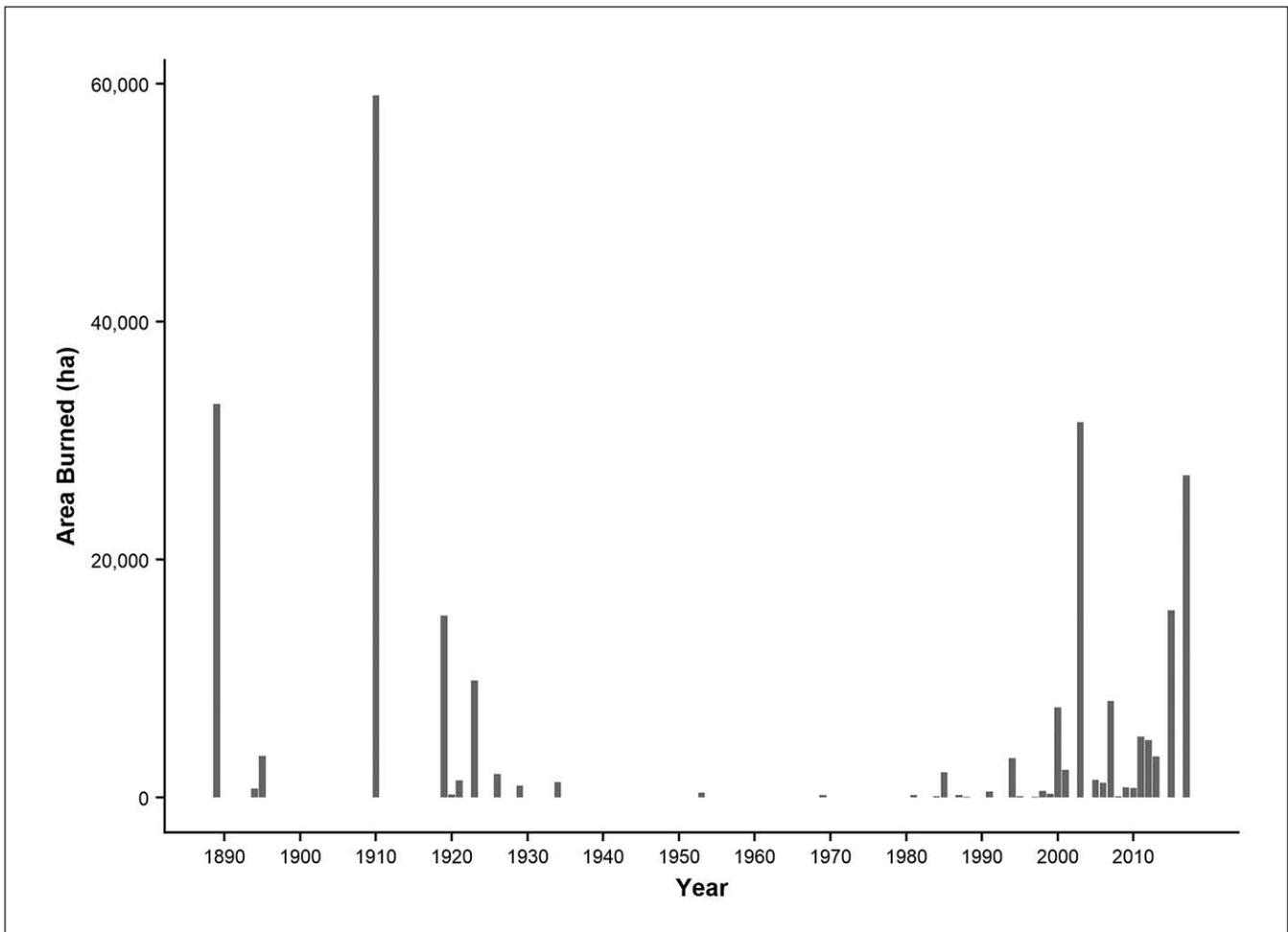


Figure 4—Area burned by year over the course of the fire atlas period. A few years (1889, 1910, 2003, 2017) account for a large percentage of total area burned, consistent with Morgan et al.’s (2008) concept of regional fire years.

Table 1—Results from spatial analyses of area that burned multiple times in the South Fork Flathead River valley (Montana) during three periods.

	Unburned	Burned 1x	Burned 2x	Burned 3x	Burned 4x
Pre-exclusion (1889–1934)					
Total area (ha)	118,386	87,147	10,375	5,701	582
% of area	53.3	39.2	4.7	2.6	0.3
Burn rate (ha yr ⁻¹)	-	1,936.6	230.6	126.7	12.9
Fire exclusion (1935–1980)					
Total area (ha)	221,606	585	0	0	0
% of area	99.7	0.3	0	0	0
Burn rate (ha yr ⁻¹)	-	13.0	0	0	0
Fire management (1981–2017)					
Total area (ha)	128,373	73,027	17,968	2,760	63
% of area	57.8	32.9	8.1	1.2	0.03
Burn rate (ha yr ⁻¹)	-	2,028.5	499.1	76.7	1.7

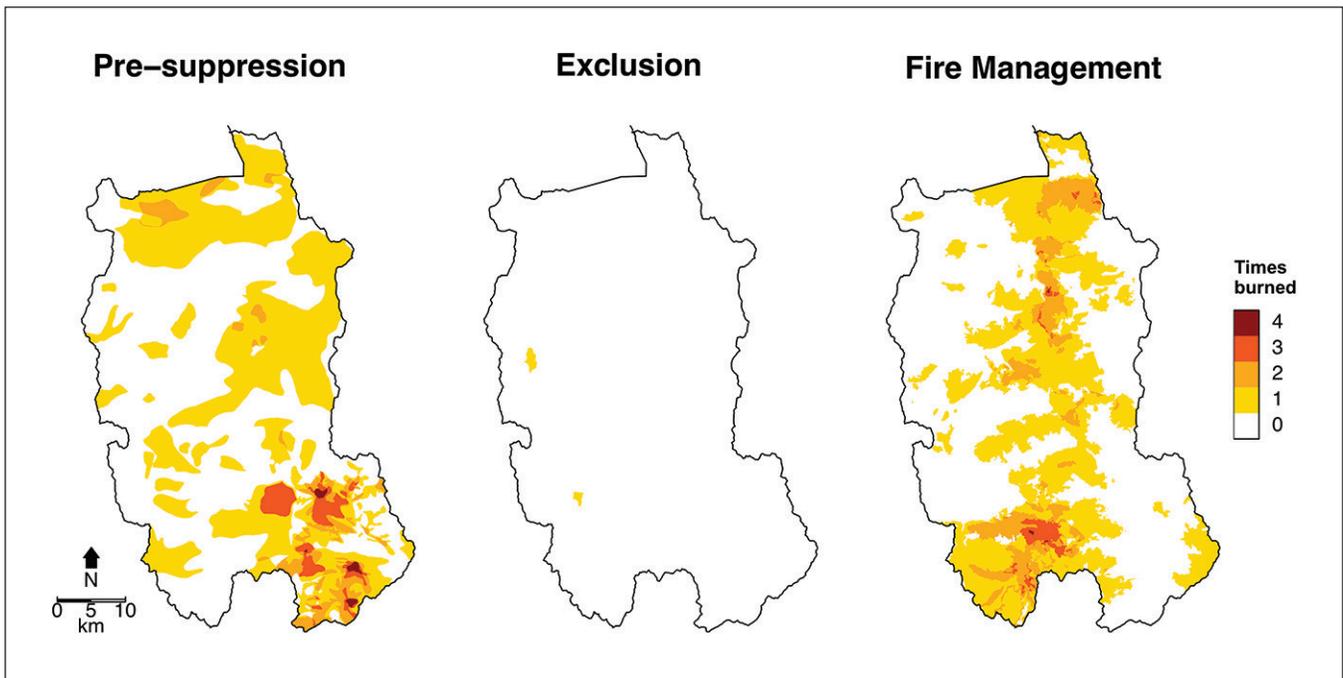


Figure 5—Areas within upper South Fork Flathead River watershed within the Bob Marshall Wilderness, Montana, USA that burned multiple times during three contiguous periods: pre-exclusion from 1889-1934 (45 years), exclusion from 1935-1980 (45 years), and fire-management from 1981-2017 (36 years).

68 years during the pre-exclusion, exclusion, and fire management periods, respectively.

Reburn Effects on Tree Regeneration, Forest Structure, and Fuels

Seedling density decreased significantly in the twice-burned plots while there was no significant decrease in once-burned plots (fig. 6, table 2). Seedling densities were not different between once-burned and twice-burned plots in either before or after periods. Sapling density significantly increased in the once-burned plots but was stable in the twice-burned plots (fig. 6, table 2). Live tree densities were stable over time in both once- and twice-burned plots. There was a marginally significant decrease in standing dead tree density in the

once-burned plots, while the twice-burned plots were stable (fig. 6, table 2).

Fine fuels in the 1 hour size class declined in twice-burned plots, with no significant decrease in the once-burned plots (fig. 7, table 3). Accumulation of 10 hour fuels was significant in the once-burned plots, while there was no change in the twice-burned plots. Hundred- hour fuels also accumulated significantly in the once-burned plots and were stable in the twice-burned plots. The large (1,000 hour) fuels were stable over time in both once- and twice-burned (fig. 7, table 3). Litter and duff depths increased significantly without fire in the once-burned plots, with no changes detected in the twice-burned plots (fig. 8, table 3).

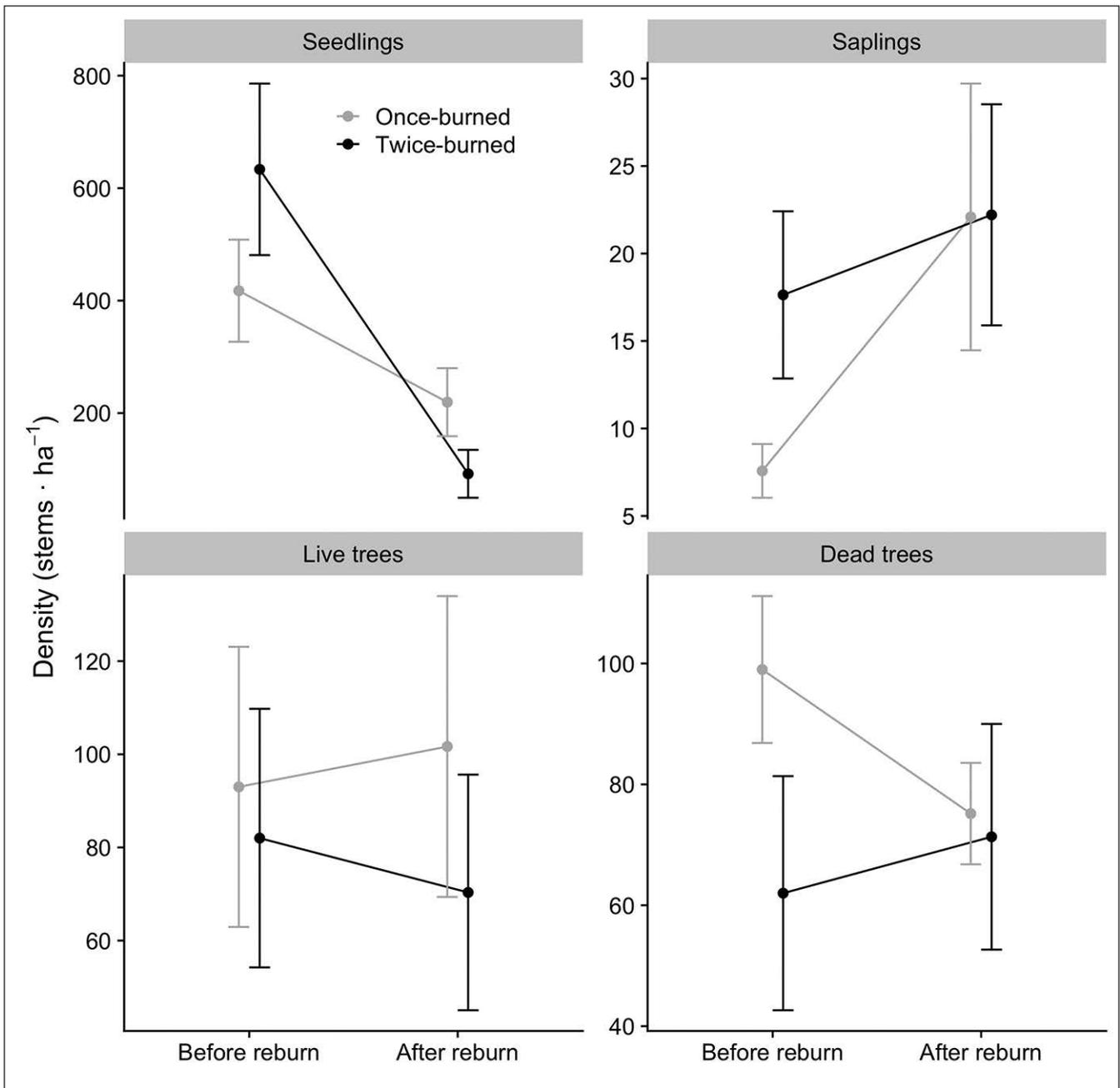


Figure 6—Effects of single and repeat fires on density of seedlings, saplings, and trees. Contrasts are between once-burned plots (2000 or 2003 fire; $n = 10$) and twice-burned plots (2013 fire; $n = 10$) and between two sampling times: before reburn (2011) and after reburn (2015). Seedlings are individuals <1.37 m tall, saplings are >1.37 m tall and <20 cm in DBH. Trees are stems ≥ 20 cm DBH. Values are means with vertical bars representing ± 1 standard error.

Table 2—Results from permutation tests on mean differences in seedling, sapling, live tree, and dead tree densities for four before-after-control-impact contrasts in burned area in the South Fork Flathead River valley (Montana). Contrasts are differences between twice-burned after reburn and twice-burned before reburn (Impact After – Impact Before; IA – IB), once-burned after reburn and once-burned before reburn (Control After – Control Before; CA – CB), before reburn twice-burned and before reburn once-burned (Before Impact – Before Control; BI – BC), and after reburn twice-burned and after reburn once-burned (After Impact – After Control; AI – AC). Significant results are indicated in bold with an asterisk. Marginally significant results are indicated in bold only.

	Contrast	Difference in density (stems ha ⁻¹)	P-value
Seedlings	IA – IB	-541	0.003*
	CA – CB	-198	0.297
	BI – BC	216	0.25
	AI – AC	-127	0.504
Saplings	IA – IB	5	0.538
	CA – CB	15	0.044*
	BI – BC	10	0.169
	AI – AC	0	0.985
Live trees	IA – IB	-12	0.776
	CA – CB	-8	0.833
	BI – BC	-11	0.778
	AI – AC	-14	0.714
Dead trees	IA – IB	9	672
	CA – CB	-24	0.287
	BI – BC	-37	0.092
	AI – AC	-4	0.866

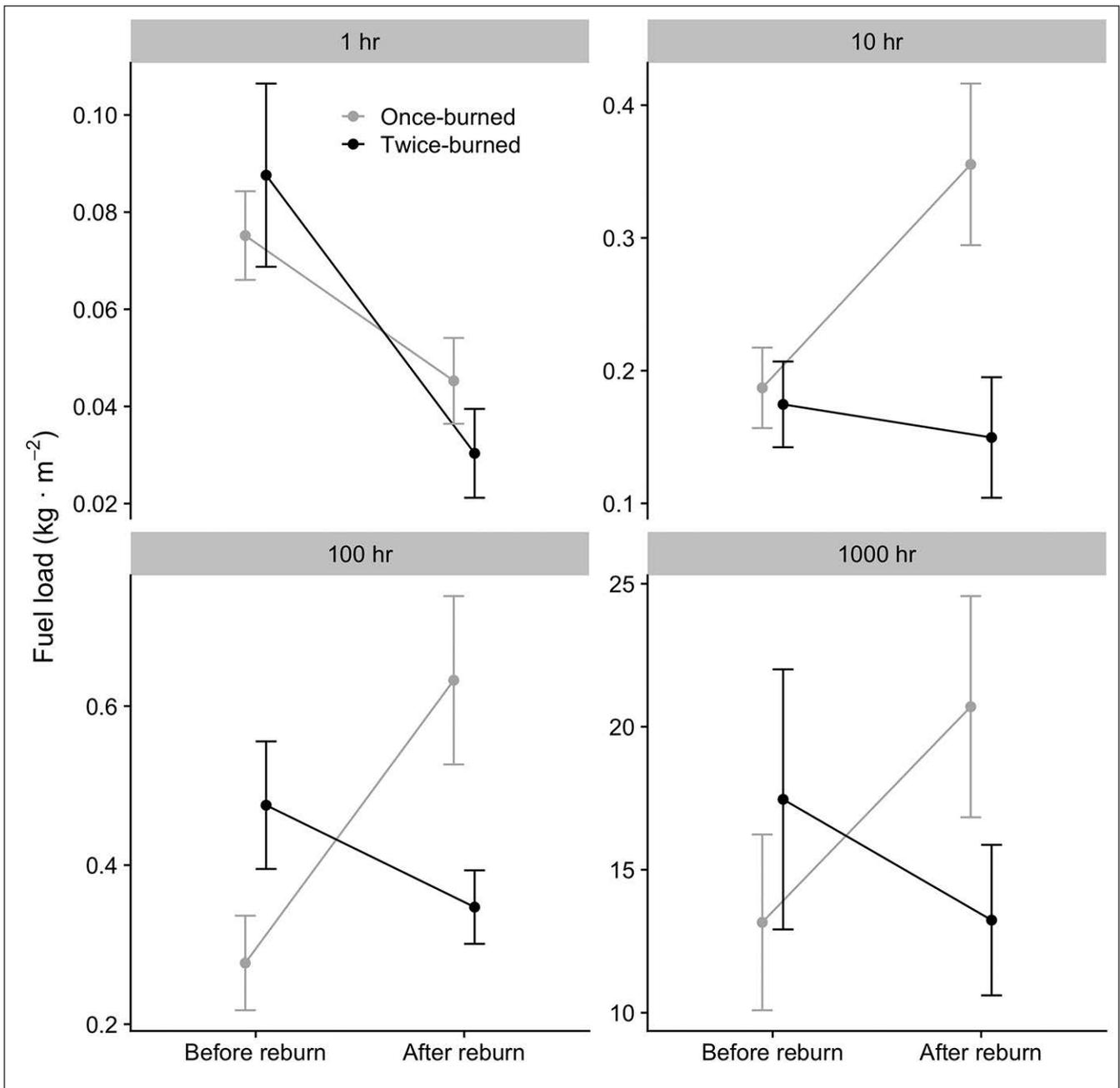


Figure 7—Effects of single and repeat fires on 1-1000 hr fuel loads. Contrasts are identical to those described in figure 6. Fine fuel (1-100 hr) loads were measured and estimated using Brown's (1974) methods. 1 hr fuels are woody debris 0-0.64 cm in diameter, 10 hr are 0.65 - 2.54 cm, 100 hr are 2.55 - 7.62 cm, and 1000 hr are >7.6 cm. Values are means with vertical bars representing ± 1 standard error.

Table 3—Results from permutation tests on mean differences in 1 hour, 10 hour, 100 hour, 1,000 hour, litter, and duff fuel amounts for four before-after-control-impact contrasts in burned area in the South Fork Flathead River valley (Montana). Litter and duff are expressed as depth (cm); 1–1,000 hour fuels as load (kg m²). Contrasts are the same as in table 1. Significant results are indicated in bold with an asterisk. Marginally significant results are indicated in bold only.

	Contrast	Difference in fuel load (kg m ² or cm)	P-value
1 hour fuels	IA – IB	-0.06	0.002*
	CA – CB	-0.03	0.141
	BI – BC	0.01	0.54
	AI – AC	-0.02	0.466
10 hour fuels	IA – IB	-0.03	0.726
	CA – CB	0.17	0.015*
	BI – BC	-0.01	0.869
	AI – AC	-0.21	0.003*
100 hour fuels	IA – IB	-0.13	0.298
	CA – CB	0.36	0.003*
	BI – BC	0.20	0.101
	AI – AC	-0.29	0.017*
1,000 hour fuels	IA – IB	-4.22	0.42
	CA – CB	7.54	0.148
	BI – BC	4.31	0.415
	AI – AC	-7.46	0.155
Litter depth	IA – IB	-0.38	0.507
	CA – CB	1.04	0.053
	BI – BC	-0.29	0.612
	AI – AC	-1.71	0.001*
Duff depth	IA – IB	-0.01	0.988
	CA – CB	1.21	0.153
	BI – BC	-0.57	0.517
	AI – AC	-1.79	0.033*

DISCUSSION

Fire History

Our analysis of 128 years of fire history data demonstrates that modern reburns have recent precedent: They were a conspicuous component of the historical fire regime in the BMW. Modern reburns are neither anomalous nor unprecedented; they are a key element of the natural fire regime in the western BMW, creating structurally and functionally distinct habitat compared to once-burned sites (Larson et al. 2013; Ward et al. 2017). The current regime of active fire since 1981 mirrors total area burned (figs. 3 and 4) and area reburned in the pre-exclusion period (fig. 5, table 1). A tree ring-based fire history in the southeastern portion of our study area also documented a regime of frequent, widespread fires, including large reburns, since 1749 (Gabriel 1976), corroborating our results and extending the temporal depth of the record with a second line of evidence.

The striking differences in total area burned and area reburned across the three time periods (table 1, fig. 5) are due to the interaction of climatic variability and fire management policy. Annual and decadal variation of climate is the primary driver of fire area burned in the northern U.S. Rocky Mountains, including the western BMW (Heyerdahl et al. 2008; Higuera et al. 2015; Morgan et al. 2008). The success of fire suppression efforts during the exclusion period (Steele 1960) was very likely conditioned upon the lower frequency of hot, dry springs and summers during the mid-20th century relative to the pre-exclusion and fire management periods, during which all regional fire years occurred in the northern U.S. Rockies (Morgan et al. 2008).

Reburn Effects on Tree Regeneration, Forest Structure, and Fuels

Reburn effects on the tree community were primarily concentrated in the smaller tree size classes: seedlings and saplings (fig. 6). Seedlings that established after the initial 2000 fire had not yet grown large enough by the second fire in 2013 to develop fire resistance traits (e.g., thick bark), and consequently suffered high mortality. Climate change may make postfire tree regeneration less successful following future fires on environmentally stressful sites (Stevens-Rumman and Morgan 2016). However, we observed abundant tree

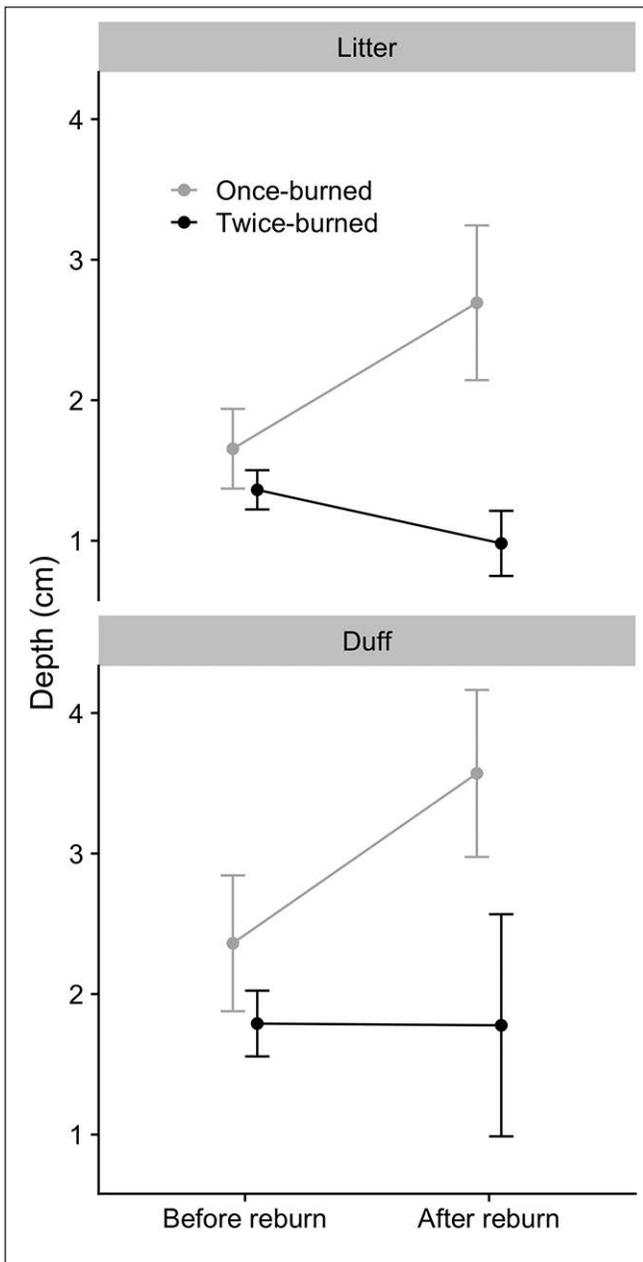


Figure 8—Effects of single and repeat fires on litter and duff fuel depths. Contrasts are identical to those described in figure 6. Litter is organic material that is finer than 1 hr fuels, but is undecomposed. Duff is partially decomposed organic material. Values are means with vertical bars representing ± 1 standard error.

regeneration establishing after both single and repeat wildfires at our sites, which were situated on the valley bottom on gentle topography. We interpret the net stability of the sapling community in the twice-burned sites as the combined effect of ingrowth of seedlings into the sapling size class balanced by fire-caused sapling mortality in the reburn event.

The overstory tree community was highly resistant to change over time in both the once-burned and twice-burned plots. The initial fires (in 2000 and 2003) preferentially removed the least fire-resistant trees through direct fire-related mortality and postfire bark beetle (family Scolytidae) attack (Belote et al. 2015b; Hood and Bentz 2007). Thus, we interpret the stability of the overstory tree population in the once-burned plots as the result of the return to low background rates of tree mortality by the time of our sampling, 8 and 12 years post-fire (Keane et al. 2006; Leirfallom and Keane 2011; Van Mantgem et al. 2011). In the twice-burned plots, the relative stability of the overstory was likely due to the high fire-resistance of the trees that survived the initial fire (Belote et al. 2015b; Harrington 2013; Larson et al. 2013), combined with modest recruitment from the sapling size class into the overstory tree size class, offsetting mortality caused by the second burn.

Single and repeat fires had sharply contrasting effects on surface fuels (figs. 7 and 8). In once-burned plots, most fuel types increased or were stable from 2011 to 2015. This reflects the ongoing deposition of bark, branches, and boles from fire-killed trees, adding to the surface fuel load (Dunn and Bailey 2012, 2015). In contrast, the second fire either reduced or maintained surface fuels in 2015 relative to 2011 levels (figs. 7 and 8). Fuel consumption in the second fire offset new deposition, leading to significant differences between once-burned and twice-burned sites in 2015 for multiple fuel classes. Based on these results, it is not appropriate to characterize single fires following a long fire-free period as “fuel reduction treatments.” Rather, single fires lead to steady accumulation of new surface fuels as fire-killed trees and branches fall to the forest floor (Dunn and Bailey 2012, 2015). In contrast, reburns do function as fuel reduction treatments, maintaining or reducing surface fuels through time (Donato et al. 2016; Stevens-Rumman et al. 2016; Ward et al. 2017).

The scope of inference for these analyses of single and repeat wildfire effects on tree regeneration, forest structure, and surface fuels is old-growth western larch/mixed-conifer forest. The presence of large-diameter, fire-resistant western larch trees is an important factor to consider when interpreting and generalizing our results (Harrington 2013). In particular, overstory stability in reburns might be diminished at sites with lesser proportions of fire-resistant species (Belote et al. 2015b). We acknowledge that this case study, while providing strong inference due to the BACI design, does not sample the full range of possible reburn effects (Coppelatta et al. 2016; Stevens-Rumman et al. 2016).

CONCLUSIONS AND MANAGEMENT IMPLICATIONS

Our analyses have implications for wilderness fire management, as well as for design of forest restoration and ecological forestry treatments outside of wilderness areas. The most important finding from our analysis of 128 years of fire history data is the similarity between the current active fire regime (1981–2017) and the pre-exclusion historical period (1889–1934), in terms of both annual area burned and amount of area burned two to four times. These results demonstrate that the modern fire regime has a recent historical precedent, and that reburns are a component of the natural fire regime of the western Bob Marshall Wilderness. Our analyses of reburn effects on tree regeneration, forest structure, and fuel loads suggest that a broader range of posttreatment conditions than described by Hopkins et al. (2014) is appropriate for combined thinning and prescribed fire treatments that seek to restore effects of past harvest and fire exclusion in western larch/mixed-conifer forests. Repeat fires result in simpler forest stand structure, lower fuel loads, and less tree regeneration than do single fires (fig. 9). The similar relative abundance of unburned, once-burned, and reburned area we observed in the pre-exclusion and fire management periods (table 1) should be informative to managers seeking to use thinning, prescribed fire, and managed wildfires to restore fire-prone forest landscapes outside wilderness areas (Hessburg et al. 2015).



Figure 9—(A) Example conditions in once-burned (in 2003) old-growth western larch/mixed-conifer forest characterized by heavy surface fuels and abundant tree regeneration (photo: AJ Larson, University of Montana). (B) Example conditions in twice-burned (in 2000 and 2013) old-growth western larch/mixed-conifer forests with reduced surface fuels and tree regeneration, and abundant charring (Ward et al. 2017) on residual coarse woody debris (photo: AJ Larson, University of Montana).

Wilderness provides value to society as a source of scientific information that enhances our ability to sustainably manage nonwilderness lands. The Wilderness Act of 1964 identifies scientific and educational uses as two of the purposes of wilderness areas. Scientists and educators are thus wilderness stakeholders who have a role in delivering to society the information value derived from wilderness areas. Managers can use the results presented here to inform the design and monitoring of forest landscape restoration prescriptions for large planning areas (*sensu* Hessburg et al. 2015), as well as for stand-level restoration (Hopkins et al. 2014) and ecological forestry (Crotteau et al. 2018) treatments that produce commercial products.

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