FINAL REPORT

Title: Influence of past wildfires on wildfire effects in northern Rockies mixed-conifer forest

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Abbreviations

ACFL – available canopy fuel load
BACI – Before-after-control-impact
BMW – Bob Marshall Wilderness
CART – classification and regression tree
CBD – canopy bulk density
CV – coefficient of variation
CWD – coarse woody debris
DBH – diameter at breast height
dNBR – differenced normalized burn ratio
LiDAR – light detection and ranging
SE – standard error
TPI – topographic position index
TWI – topographic wetness index

Keywords

Fire effects, fire ecology, western larch, surface fuel, canopy fuel, forest structure, reburn, wilderness, wilderness fire, Bob Marshall Wilderness

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Abstract

Natural resource managers need to know how past wildfires influence the severity and ecological effects of subsequent wildfires fires in order to make informed decisions during and after wildfire events, and to effectively plan for the future. The overarching goals for this study were to quantify and compare the effects of single and repeat wildfires on forest ecosystem structure and fuels; to determine if past wildfires influence the severity of subsequent wildfires; and to determine if short-interval reburns are causing transitions to non-forest communities.

Since the early 1980s managers have allowed many lightning-ignited fires to burn with minimal interference in forests of the Bob Marshall Wilderness in northwestern Montana, USA. Substantial portions of this landscape burned twice between 1985 and 2013. We used this active fire regime to investigate fire-effects and post-fire fuel loads, tree regeneration, and forest structure in mixed-conifer forest communities. We used an extensive network of field plots (n = 264 total plots) distributed among three sampling protocols to meet our objectives.

Fire history strongly affected forest structure and fuels. Surface and canopy fuels exhibited contrasting responses, with the surface fuel complex buffered by post-fire inputs from the overstory. A second fire is required to cause meaningful reductions to the surface fuel loads, and three or more fires may be required to reduce the largest (≥1000 h) fuels, except in situations where high severity patches are reburned. Fire effects on canopy fuels are much more predictable, with steady reductions of canopy fuels, tree biomass, and total aboveground biomass along the fire history gradient from unburned, to once-burned, to reburned sites. Live tree density was best explained by an interaction between initial fire severity and topography: live tree density was lowest on steep southerly aspects that burned in at high severity. Environmental variables related to topographic position and the severity of the initial fire, but not necessarily the occurrence of a reburn, were important in explaining transitions to non-forest following fire.

Our most important finding is that surface fuel loads are maintained or increased in the years following an initial wildfire after a long fire-free period as fire killed trees and branches fall to the ground. This unsurprising result nevertheless deserves highlighting because the current conventional wisdom is that an initial fire can be thought of as a "fuel treatment." Our most surprising finding was the unimportance of reburns as a cause of transitions to non-forest.

We maintained a very active and successful science delivery and outreach program during this project. Science delivery activities included eight scientific, workshop, and public presentations; two completed and two in preparation publications; and multiple media contacts and interviews resulting in this research being featured in two different news articles, a book, and a short documentary film. This diversified science delivery program reached managers, scientists, students, and the general public through multiple platforms.

Managers need to plan for multiple fire entries (i.e., two or more fires) if their goal is to use wildfires as surface fuel reduction treatments. Our results demonstrate that some transitions to a putative non-forest condition are to be expected following both initial fires and short-interval reburns. Thus, managers may wish to incorporate this outcome into their expectations, and into their outreach and education efforts, in order to prepare policy makers and the public for forest conversion. Numerous historical reconstructions have shown that many formerly fire-maintained open areas have been encroached by forest during the period of fire exclusion—returning some areas to a putative non-forest condition may actually be restorative from a landscape perspective.

Objectives

Our overarching goals were to determine if past wildfires influence the severity of subsequent wildfires; to quantify and compare the effects on forest ecosystem structure and composition of single and repeat wildfires; and to effectively communicate our findings to natural resource managers. In particular, we ask if the ultimate effects on forest structure, composition, and fuels of short interval "reburns" depend on the severity of the initial fire (reburns are defined here as the two fires occurring in the same place within 30 years).

Since the early 1980s managers have allowed many lightning-ignited fires to burn with minimal interference in forests of the Bob Marshall Wilderness in northwestern Montana, USA. Substantial portions of this landscape burned twice between 1985 and 2013. We used this active fire regime to investigate the comparative effects of single and repeat wildfires on fuel loads, tree regeneration, and forest structure in mixed-conifer forests, including transitions to non-forest.

Objective 1. Quantify the effects of reburns on fuels, forest structure, and tree populations in western larch/mixed-conifer forests. We hypothesized that reburns will: (1) reduce surface fuels relative once-burned sites levels, (2) reduce tree seedling and sapling (<20 cm dbh) densities relative to densities at once-burned sites, and (3) cause little mortality of overstory trees (>20 cm dbh) that survived the initial fire. These straightforward predictions were consistent with our earlier work in the Bob Marshall Wilderness in ponderosa pine/mixed-conifer forests (Larson et al. 2013). Nevertheless, our initial findings required validation because managers are presently forced to base and justify fire management decisions in this forest type to an uncomfortably large degree on assumptions and anecdote.

Objective 2. Determine if the severity and effects of reburns depend on the severity of the initial fire, and if reburns are causing shifts to non-forest communities. An unresolved issue in the literature is if, and under what circumstances, the severity of reburns are moderated (e.g., Parks et al. 2013) or enhanced (e.g., van Wagtendonk et al. 2012) by the initial fire. In our earlier case study (Larson et al. 2013) we found evidence that the severity of the initial fire influenced the outcomes of the reburn. In places where the initial fire was severe (leading to locally dense accumulations of coarse surface fuels), the second fire burned with relatively greater intensity and severity than locations that burned with low or moderate severity in the initial fire (Larson et al. 2013). We hypothesized that in locations where the initial fire burned with low or moderate severity, reburn effects will be moderated (stabilizing feedback), maintaining a low-density, multistory forest. But, in locations where the initial fire burned with high severity, reburn effects on the forest community will be exacerbated (amplifying feedback), creating a structurally and compositionally simplified forest (Figure 1), or even causing a transition to a non-forest community. We expect this threshold effect—a switch from a stabilizing to amplifying feedback—with increasing fire severity is due to the prolonged burning of heavy accumulations of coarse surface fuels arising from high tree mortality levels in the initial fire. This mechanism differs from that postulated for Sierra Nevada forests, where initial high severity fires can cause a transition to a more flammable montane chaparral community, which amplifies fire severity in reburns (van Wagtendonk et al. 2012). The effects of reburns may also depend on topographic setting (Arno et al. 2000, Lydersen and North 2012).

Background

Ecosystems can shift to alternative stable states following disturbance, changing environmental conditions, and the interactions between disturbance and changing conditions (Beisner et al. 2003). If stabilizing feedbacks that maintain variability within a range of conditions are disrupted, an ecosystem can shift to an alternative state (Chapin et al. 1996). The alternative state can then be maintained by new stabilizing feedbacks. For example, forests can convert to persistent shrub or herbaceous-dominated ecosystems following uncharacteristic fire (Fletcher et al. 2013). The shrub or herbaceous ecosystems are then maintained through a new fire regime. Grasslands can also convert to shrublands via intense and persistent grazing, which can subsequently be maintained by competition for belowground resources (Bestelmeyer 2006). Understanding these possibilities has important implications for management, restoration, and sustainability of ecosystem services and biological diversity.

Fires that burn over relatively recently burned areas, called reburns, may induce state shifts by killing regenerating species or by modifying the physical (coarse or fine woody debris) or chemical (nitrogen pools) characteristics of sites. For instance, fire can result in a new regeneration cohort of seral species. If a fire reburns new seedlings before reproductive age, a site may be depleting of seed sources ultimately eliminating trees from a site for long periods.

Figure 1. (Left) Example of an area that burned at low severity in 2003 and 2011, suggesting that fire effects are self-limiting (stabilizing feedback) when the initial fire burns at low severity. (Right) Example of an area (foreground) that burned at high severity in 2003 and 2011, suggesting that fires increase reburn severity (amplifying feedback) when the initial fire burns at high severity.

Natural resource managers need to know how past wildfires influence the severity and ecological effects of subsequent wildfires fires in order to make informed decisions during and after wildfire events, and to effectively plan for the future (North et al. 2012). If the preponderance of scientific evidence indicates that past wildfires moderate the severity of subsequent fires—essentially functioning as fuel treatments—managers may use different fire management approaches if most evidence indicates that past fires tend to increase severity of subsequent fires.

Parks et al. (2014) recently argued that wildfires can be considered effective fuels treatments. They reached this conclusion based on their finding that previous fires moderated the severity of subsequent fires (Parks et al. 2014), and on the finding of Teske et al. (2012) that previous fires limited the spread of subsequent fires. Results from these studies must be interpreted with caution because they are based on indices of burn severity derived from satellite imagery, not measurements of post-fire community composition, structure, and fuels. Indeed, much of the reburn research available in the published literature disproportionately relies on remotely sensed data (e.g., Thompson et al. 2007, Collins et al. 2009, Teske et al. 2012, van Wagtendonk et al. 2012, Parks et al. 2014). Remotely sensed data offer many benefits, for example the ability to study very large areas or to answer research questions that are intractable using field based approaches alone (e.g., Cansler and McKenzie 2014). But, if remotely sensed metrics are not clearly related to ecologically meaningful field-based measurements of fire effects they offer little value to managers, and may even be misleading. Detailed field measurements of unburned, once-burned, and reburned areas are needed to identify if, and under what circumstances, wildfires function as fuel treatments by moderating severity (Parks et al. 2014) of subsequent fires, or, alternatively, increase the severity (van Wagtendonk et al. 2012) of subsequent fires.

Materials and Methods

Study Area

The study area comprises the lower slopes and valley floor of the portion of the upper South Fork Flathead River watershed (above Bunker Creek) within the BMW, Montana, USA. Elevation along the main stem of 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 (*Larix occidentalis*), Engelmann spruce (*Picea engelmannii*), and subalpine fir (*Abies lasiocarpa*), with minor amounts of ponderosa pine (*Pinus ponderosa*) (Belote et al 2015).

Field Methods

We used three different field sampling protocols in this study. The first was a Before-After-Control-Impact (hereafter, "BACI study") design comprising n = 20 plots that were measured in 2011 before the initiation of this study, then measured again in 2015 after half the plots reburned. The second was an independent sample of n = 10 once-burned and n = 10 twice-burned locations which we used to investigate stocks of black carbon on CWD in once-burned and twice-burned sites (hereafter, "black carbon study"). The third was a network of n = 224 unburned, once-burned, and reburned sites (hereafter, "UOR study"), which we used to investigate fire effects on forest structure and fuels; relationships between severity of the first and second fires; and transitions to non-forest communities. We report the methods and results/discussion from these three different sampling protocols in a parallel structure for clarity.

Field Methods: BACI study

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 (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. The area west of the river did not burn a second time. All three fires were ignited by lightning.

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. 2015). Half of these plots reburned in the 2013 fire. In 2015, all plots were relocated using GPS 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 a 1 m radius subplot at plot center. To inventory saplings (>1.37 m tall and <20 cm diameter at breast height [DBH]), 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 DBH), we recorded the species, diameter, tree type (live standing tree, dead standing tree, or uprooted and/or snapped below DBH but inferred standing at time of fire) within 17.84 m of plot center. We recorded trees with a diameter at breast height 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; Van Wagner 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 hr (0-0.64 cm) and 10 hr (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 hr 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 ≥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 Methods: black carbon study

We located n = 10 sites in once-burned areas (five in 2000 fire only and five in 2003 fire only), and n = 10 sites in twice-burned areas (five in 2013 reburn of 2003 fire and five in 2011 reburn of 2003 fire). Sample sites were chosen randomly from patches within the initial fire that was classified as high-severity burn and at least 3 x 3 pixels (90 m × 90 m) in area, using burn severity maps from the Monitoring Trends in Burn Severity program (mtbs.gov).

CWD and associated charcoal were sampled in August 2014 using the planar intercept method (Donato et al. 2009). Sampling transects were arranged in a 30 m × 30 m square oriented to the cardinal directions and along one interior diagonal of the square, for a total of 162.4 m of transect per site. We recorded diameter, species, decay class, and depth of char for each CWD piece that intersected the sampling plane.

Field Methods: UOR study

Between 2015 and 2017, we sampled 224 vegetation plots within the Bob Marshall Wilderness (Figure 2). We sampled sites that had not burned since at least 1935 (unburned), sites that had burned once between 1985 and 2013 (once-burned), and sites that had burned twice between 1985 and 2013 (reburned). Plot locations were selected using a stratified random sampling design. To ensure that we sampled a full range of burn severity and fire condition across topographic gradients, we stratified our plots by a factorial combination of fire (unburned, once burned, and reburned), initial fire severity, and topographic classes. We used the fire atlas produced by Parks et al. (2015) to identify unburned regions, once burned regions, and reburned regions in the study area. Because the Parks et al (2015) fire atlas did not include dNBR or perimeter data from the Damnation fire of 2013 we downloaded data for this fire from MTBS.gov and added it to the fire atlas. Burn severity of all data were classified into four classes. We used the year of fire attribute data to calculate years between fires and added this information to the attribute table of the reburned polygons. We used a 30-m DEM to calculate slope and aspect within the study area. We then created three topographic classes (flat, northeast-facing, and southwest-facing) based on each pixel's slope and aspect.

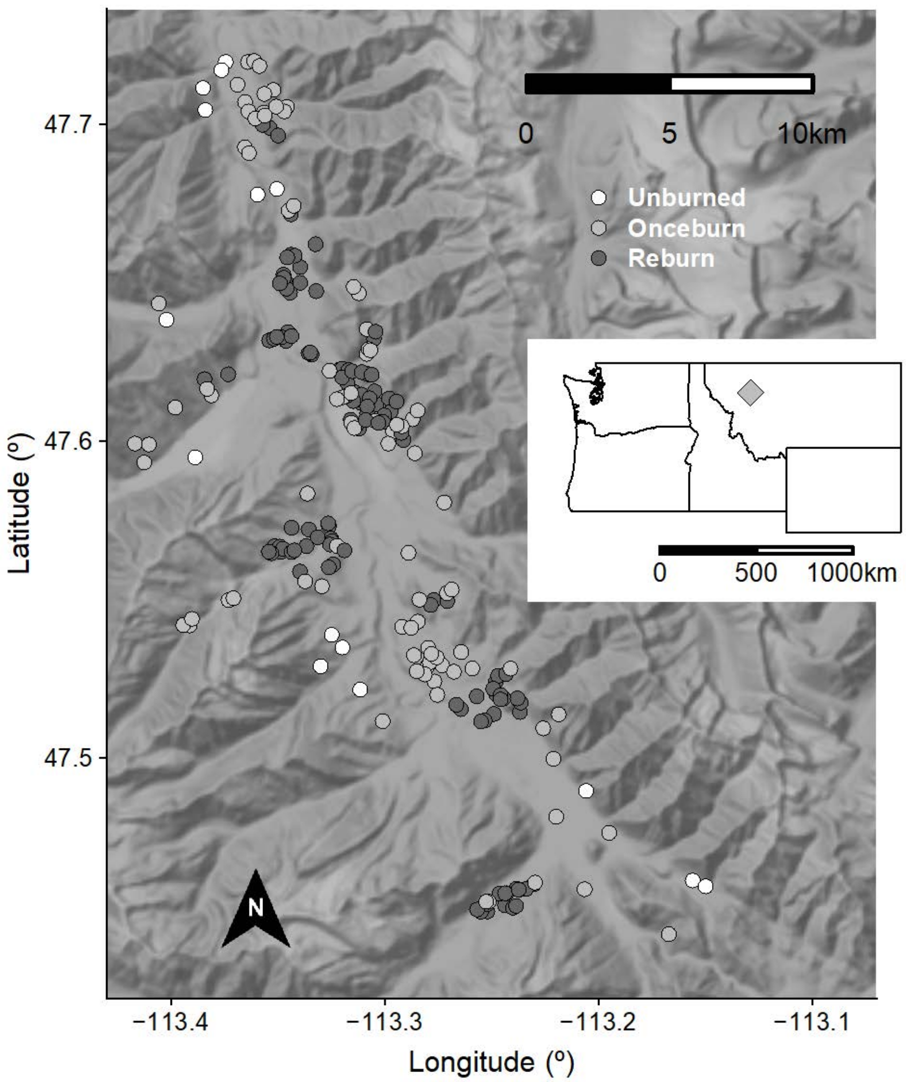

Figure 2. Map showing field plot locations from the Unburned-Onceburned-Reburned comparative study in the South Fork Flathead River watershed, Bob Marshall Wilderness, Montana, USA. Different color dots represent unburned (white; *n* = 15), once-burned (light grey; *n* = 89), and reburned (dark grey; *n* = 120) plots.

In each plot, we measured surface fuels along three 20 meter Brown's transects (Brown 1974). The first transect ran directly east from plot center, the second transect was located at a 330 degree azimuth beginning from the end of the first transect, and the third transect was at a 270 degree azimuth from the end of the second transect. 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 10 m to 13 m from plot center. We similarly tallied the number of intersections of 100 hour fuels (2.55 - 7.62 cm) from 10 m to 16 m from plot center. All coarse woody debris (≥7.62 cm) that intersected the transects were counted as well, and for these larger fuels we measured the diameter at the point of intersection as well as the decay class (1-5, with 1 indicating a sound log with no decay and 5 indicating a very decayed log). At the 10 and 20 meter mark along each transect, we measured litter (undecomposed organic material) and duff (partially decomposed organic material) depths.

We conducted visual estimates of live and dead shrub and herb fuel loads using photoload microplots according to the methods of Keane and Dickinson (2007). For these microplots, a 1 m x 1 m sampling area was established at 0, 10, and 20 meters along each transect for a total of 7 microplots per plot. For each microplot, we recorded visual estimates of shrub and herb fuel loading, estimates of shrub and herb cover (live and dead), as well as an estimate of both shrub and herb canopy height. The fuel loading estimates were based off of pictures provided in the appendices and training documents of Keane and Dickinson (2007).

At each plot we also collected data on tree seedlings and saplings. Seedlings (< 1.37 meters tall) were tallied by species within a 4 m x 20 m plot, centered on the first Brown's transect oriented at a 90 degree azimuth from plot center. Within this plot, all live tree seedlings were counted and recorded in 3 height classes: 0-40 cm, >40-80 cm, and >80 cm-137 cm. Saplings were sampled within a 5.00 m or 17.84 radius plot, depending on whether the site had been unburned or burned, respectively. Within the sapling plot, live and dead saplings, respectively, were counted by species and dbh class: 0-2.5 cm, 2.5-5 cm, 5-7.5 cm, and 7.5-10 cm. Average height and live crown percentage was also recorded for each sapling size× species × status category present in a plot.

All live and dead standing trees with ≥10-80 cm dbh were individually measured within a 17.84 m radius plot (0.1 ha); trees > 80 cm dbh were sampled within a concentric 43.7 m radius (0.6 ha) plot. We recorded tree species, status (live/dead), dbh, total height, height to base of crown, and live crown percent. In addition, we recorded whether or not the tree had a broken top. For dead trees, we additionally collected data on the decay class of the tree (1-5, with 1 meaning all barks and limbs remains and 5 meaning most of the bark and branches had been lost), the amount of foliage remaining, and whether or not the inner tree had been charred.

We extracted a variety of spatial data representing fire severity, topographic position, and climatic conditions to each plot location using bilinear interpolated values. We extracted data on dNBR from all fires that burned plot location since 1984, time between fires, number of times burned, elevation, slope, aspect, a topographic wetness index, a heatload index, a topographic position index, potential evapotranspiration, actual evapotranspiration, and average annual moisture deficit. The heatload index is metric of the potential radiation load of a site based on its aspect, slope, and latitude (McCune and Keon 2002). Higher heatload index values represent relatively sunnier sites. Topographic wetness index (TWI) is a modeled metric of potential water flow accumulation in a landscape estimated from the upslope contributing area and slope (Beven and Kirkby 1979). Normalized topographic position index (TPI) is a measure of landform shape. Specifically, normalized TPI measures the topographic position as a fraction of local relief scaled by local surface roughness (i.e., the standard deviation of elevation of surrounding pixels; De Reu et al. 2013). Positive values indicate convexities, negative values concavities.

Data Reduction and Analysis: BACI Study

We summarized fine fuel (1-100 hr) loads for each plot using Brown's (1974) equations for mixed-species fuels. We classified all CWD as 1000 hr fuels. To estimate 1000 hr 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. 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 (1-1000 hr), and litter and duff depths.

Data Reduction and Analysis: black carbon study

Charcoal mass estimation involves first making the standard planar intercept CWD volume calculation for each CWD piece including the charred rind, as well as calculating the volume of the inner uncharred core by reducing CWD piece radius by the measured char depth (Donato et al. 2009). The difference of these two cylinders is the volume of charcoal on the CWD piece. We calculated the total CWD volume using Eqn. 1 in Donato et al. (2009) and bias-corrected charcoal volume using Eqns. 1, 3, and 8 in Donato et al. (2009). We converted CWD volumes to mass estimates using species and decay class specific CWD densities (Bisbing et al. 2010), and estimated black C mass using Eqn. 4 in Donato et al. (2009). We tested for differences of black C mass (kg ha^{-1}) and total CWD biomass (kg ha^{-1}) between once-burned and twice-burned forests using two-sample Wilcoxon rank sum tests. All analyses were performed in the R environment.

Data Reduction and Analysis: UOR study

We estimated surface fuel loads of 1-1000h fuels using Brown's (1974) methods. To estimate litter and duff loads, we first averaged depths among the 6 measurements per plot (3 transects with 2 measurements each) then multiplied by a bulk density of 148.696 $kg \cdot m^{-3}$ (derived from regional estimates by Keane et al. 2012). Shrub and herb loadings were estimated in the field using the photoload sampling technique (Keane and Dickenson 2007).

We estimated aboveground biomass of trees, saplings, and seedlings using previously published species-specific allometric equations. We used Brown's (1978) equations to estimate tree crown biomass for trees and saplings and to estimate whole tree biomass for trees, saplings, and seedlings less than 4.6 m tall. We used the subset of Brown's equations that predict biomass from dbh and/ or tree height. For sapling and tree stems, we either used Brown's (1978) equations (stems < 10 cm dbh) or Jenkins et al. (2003) equations (stems > 10 cm dbh). Tree, sapling, and seedling biomass was split into foliage, 1-100 h fuels (crown branches and small stems), and 1000 h fuels (large stems) based on Brown's and Jenkins proportions, as appropriate. Tree crown biomass was estimated for each tree while sapling and seedling biomass was estimated in groups using field counts of saplings and seedlings in each dbh×height class.

Because the allometric equations provide estimates for intact live trees and saplings, we incorporated biomass losses for burnt trees, dead trees, and trees with broken tops. Specifically, we used field estimates of foliage retention and standing decay class (Harmon et al. 2008) to derive reduction factors for foliage and woody fuel component (1-100 h) mass of crowns. Needle retention estimates were grouped into four bins - 100%, 75%, 25%, and 0% - which corresponded with reduction factors of 0, 0.25, 0.75, and 1 that were multiplied by Brown's foliage biomass estimates, respectively. Following Donato et al. (2013), we assumed 0% loss of 1h crown fuels for standing decay class 1 and 100 % loss for all higher decay classes. Crown biomass in the 10 h and 100 h fuel classes were reduced 0%, 20%, 50%, 80%, or 100% for standing decay classes 1-5, respectively. To estimate the loss of crown biomass for broken top trees, we first regressed height by dbh using a generalized additive model for all the non-broken top trees (n = 1878). This resulted in a curvilinear relationship with R^2 equal to 0.765 ($P < 0.001$). We then used this relationship to predict the unbroken height of broken top trees. The inverse of the ratio of measured broken height to modelled height was taken as a reduction factor for crown biomass of trees with broken tops (Donato et al. 2013). We used Harmon et al.'s (2008) density reduction factors to estimate biomass change in 1000 h biomass for decay classes 1-4 (classes 4 & 5 were assumed to have the same density reduction) in standing dead stems. We applied another reduction factor to stem biomass in large trees with broken tops by calculating the ratio of the volume of two cones representing the lost stem section and the entire unbroken stem. We used the ratio of two cone volumes instead of ratios of 1-dimentional heights to more accurately account for the effect of taper on stem biomass. The denominator was calculated as the volume of a cone with base diameter equal to measured dbh with height equal to modelled unbroken height. The numerator was then the volume of the smaller cone with height equal to modelled unbroken height minus the measured broken height.

After crown biomass reductions were applied, the available canopy fuel load (ACFL) contribution for each tree and sapling was calculated as a sum of its live and dead foliage and 1h crown fuel (50% of live 1h and 100% of dead; Reinhardt et al. 2006) divided by the relevant plot area (e.g., 6000 m^2 for trees > 80cm dbh, 1000 m^2 for smaller trees). ACFL contributions were then summed for each plot. We estimated canopy bulk density (CBD) as the maximum of a 3 m running mean of the vertical ACFL profile for each plot (Reinhardt et al. 2006). ACFL profiles for each plot were calculated as summed values of each tree and group of saplings and seedlings by 0.25 m bins from the ground to the tallest tree in the plot. Seedling and sapling profiles were assumed to be evenly distributed along their entire height. Tree ACFL profiles were assumed to be evenly distributed from crown base height to the top of the tree. Crown base height was measured in the field for live trees and estimated for dead trees using a generalized additive model predicting crown base height with dbh for live trees (n = 1949 live trees, $R^2 = 0.33$, $P < 0.001$). Dead trees typically had small contributions to ACFL profiles after fine crown fuels were reduced according to decay class.

To test the significance of observed differences in mean fuel and biomass variables between unburned, once-burned, and reburned plots, we compared observations against estimated sampling distributions generated from 100,000 random permutations of unburned, once-burned, and reburned plot labels. A two-sided P-value was calculated for each test as the proportion of permuted differences in means whose absolute values were greater than the observed difference. All analyses were made in the R environment.

We used a linear model in a multiple regression framework to investigate relationships between post-fire live tree density and fire severity, time since fire, and environmental conditions, as well as their interactions. We included number of fires, years between fires, severity of the initial fire and reburn severity, topographic position index, topographic wetness, heatload, slope, aspect, elevation, and water deficit as well as two-way interactions between all variables into a full model as potential predictors of live tree density.

We subset plots that experienced reburns (n = 120) and evaluated the relationship between initial fire severity with the reburn severity using dNBR data extracted to plot locations. We quantified the nature of the relationships using quantile regression and fit models to the 10^{th}, 50^{th}, and 90^{th} percentiles of the data (Cade and Noon 2003).

We used classification methods to identify putative non-forest states following wildfires and machine learning techniques to characterize biophysical settings where state shifts may have occurred. Specifically, we classified plots into putative states based on the number of live seedlings, saplings, and trees remaining after at least one wildfire. Then, we used machine learning techniques (e.g., classification and regression trees (CART) and random forest approaches) to identify biophysical conditions where shifts to putative non-forest states occurred.

To begin, we used simple rules to classify plots into two groups: those without any live trees (seedlings, saplings, and larger trees) and those plots with live trees. From this simple classification, we evaluated the environmental conditions that characterize locations of non-forest states following fire and the species composition of living and dead trees associated with forest and non-forest classes. We used a rank of variable importance scores from random forest models to identify the environmental variables that best discriminate between the forest and non-forest states (Cutler et al. 2007). We also classified plots based on density of live seedlings (plots with no live seedlings were classified as such) and re-ran and random forest model. We conducted this analysis on the full dataset to evaluate conditions where putative state shifts to non-forest conditions occurred across all plots, and then subset the data to the reburned areas to investigate whether similar environmental variables explained putative shifts.

Results and Discussion

BACI study

Seedling density decreased significantly in the twice-burned plots while there was no significant decrease in once-burned plots (Figure 3, Table 1). 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 (Figure 3, Table 1). 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 (Figure 3, Table 1).

Table 1. Results from permutation tests on mean differences in seedling, sapling, live tree, and dead tree densities for four BACI contrasts. 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	0.779
	CA - CB	-9	0.833
	BI - BC	-11	0.787
	AI - AC	-31	0.434
Dead trees	IA - IB	9	672
	CA - CB	-24	0.287
	BI - BC	**-37**	**0.092**
	AI - AC	-4	0.866

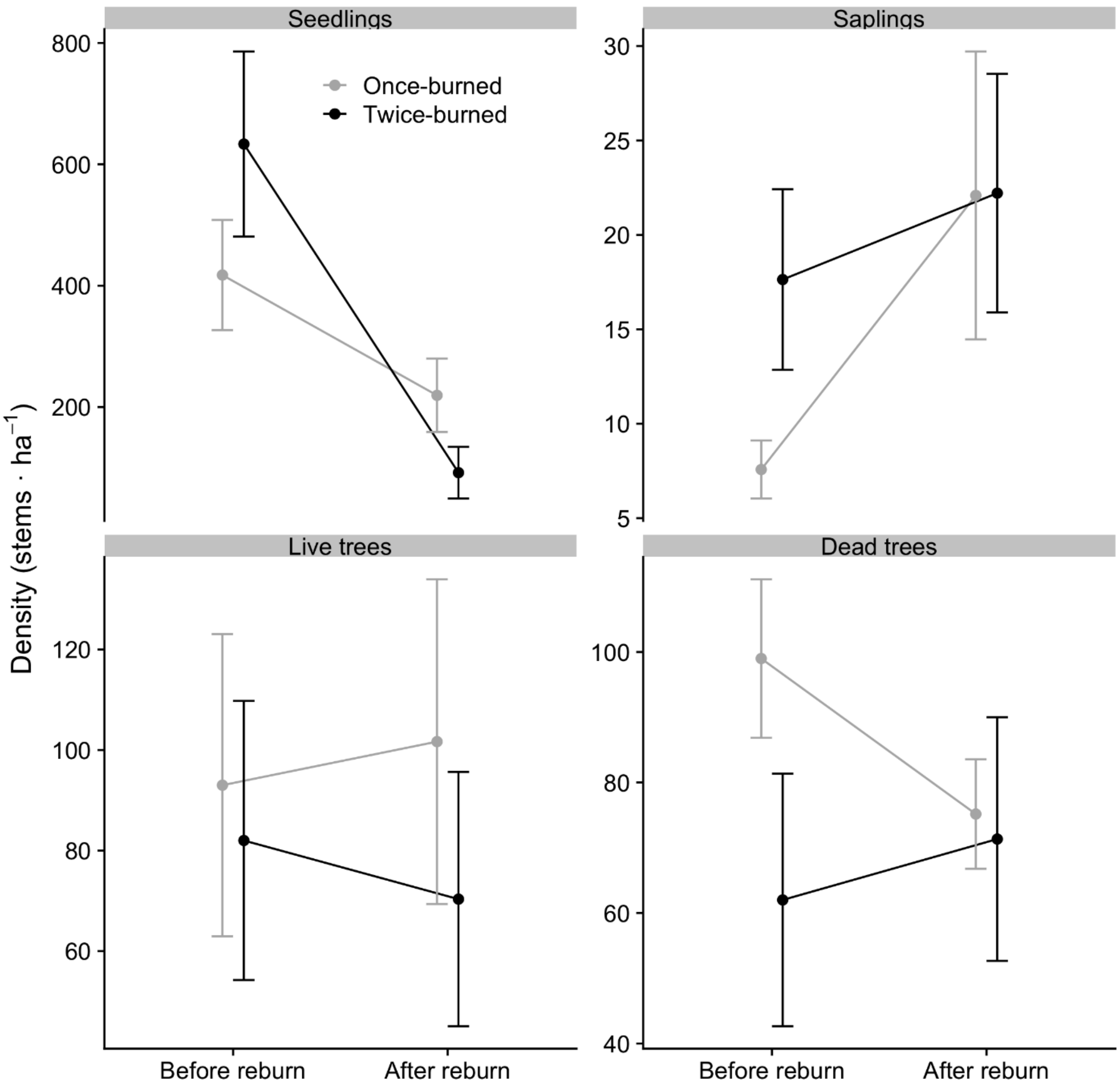

Figure 3. Effects of single and repeat fires on density of seedlings, saplings, and trees. Contrasts are between once-burned plots (2000 or 2003 fire; *n* =10) and twice-burned plots (2013 fire; *n* = 10) and between two sampling times: before reburn (2011) and after reburn (2015). Seedlings are individuals <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.

Reburn effects on the tree community were primarily concentrated in the smaller tree size classes: seedlings and saplings (Figure 3). 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 post-fire tree regeneration less successful following future fires on environmentally stressful sites. However, we observed abundant tree 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 post-fire bark beetle attack (Hood and Bentz 2007; Belote et al. 2015). 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; Lierfallom and Keane 2010; 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 (Harrington 2013; Larson et al. 2013; Belote et al. 2015), combined with modest recruitment from the sapling size class into the overstory tree size class, offsetting mortality caused by the second burn.

Fine fuels in the 1 hr size class declined in twice-burned plots, with no significant decrease in the once-burned plots (Figure 4, Table 2). 10 hr fuels accumulated significantly in the once-burned plots, while there was no change in the twice-burned. 100 hr fuels also accumulated significantly in the once-burned plots and were stable in the twice-burned plots. The large 1000 hr fuels were stable over time in both once- and twice-burned (Figure 3, Table 2). Litter and duff depths increased significantly without fire in the once-burned plots, with no changes detected in the twice-burned plots (Figure 5, Table 2).

Single and repeat fires had sharply contrasting effects on surface fuels (Figures 4 & 5). 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 (Figures 4 & 5). 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’ with respect to woody surface fuels. Rather, single fires lead to steady accumulation of new 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).

Table 2. Results from permutation tests on mean differences in 1 hour, 10 hour, 100 hour, 1000 hour, litter, and duff fuel amounts for four BACI contrasts. Litter and duff are expressed as depth (in cm), 1-1000 hour fuels as load (in kg·m^2). 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***
1000 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***

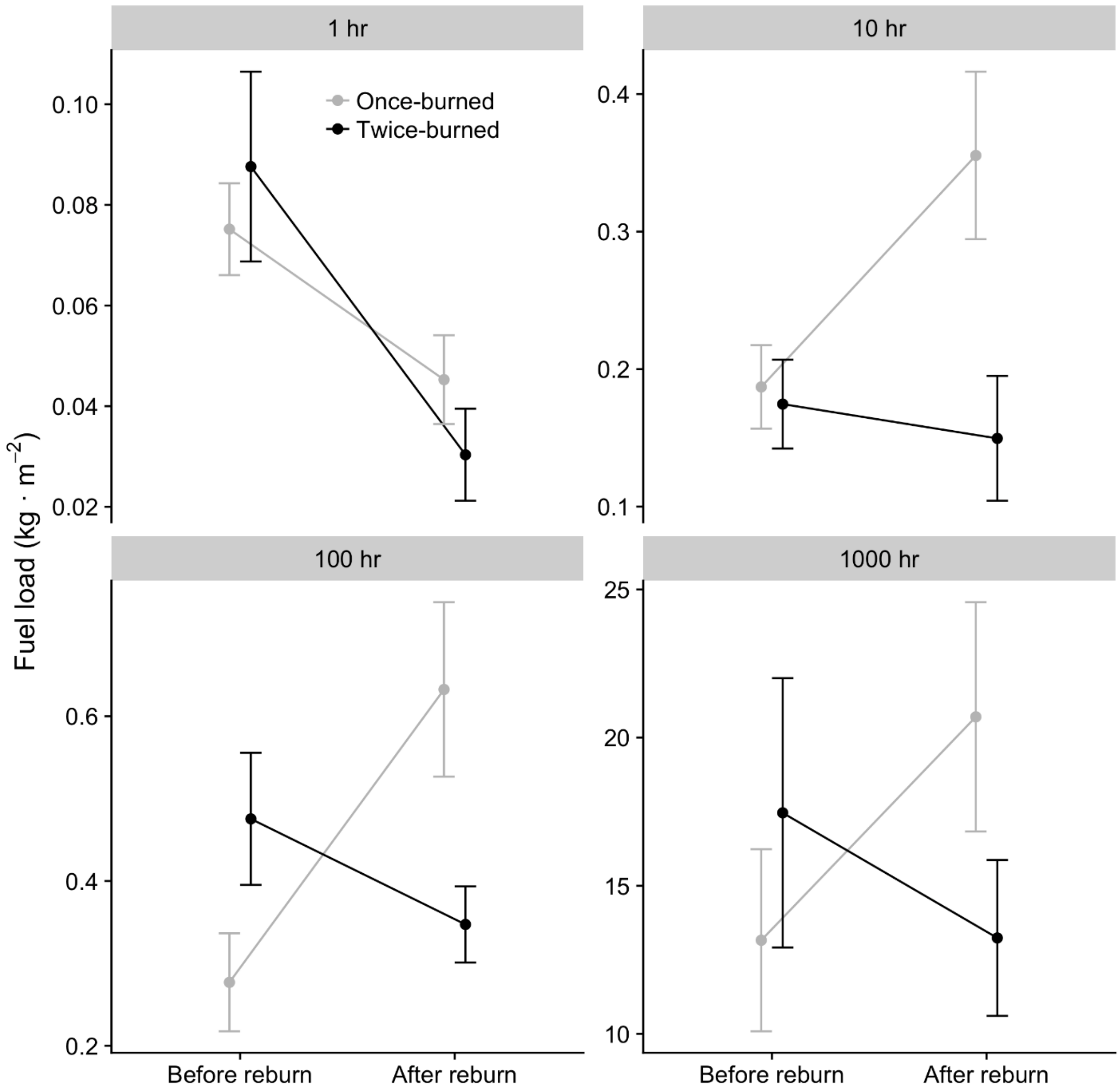

Figure 4. Effects of single and repeat fires on 1-1000 hr fuel loads. Contrasts are identical to those described in figure 6. Fine fuel (1-100 hr) loads were measured and estimated using Brown's (1974) methods. 1 hr fuels are woody debris 0-0.64 cm in diameter, 10 hr are 0.65 - 2.54 cm, 100 hr are 2.55 - 7.62 cm, and 1000 hr are >7.6 cm. Values are means with vertical bars representing ±1 standard error.

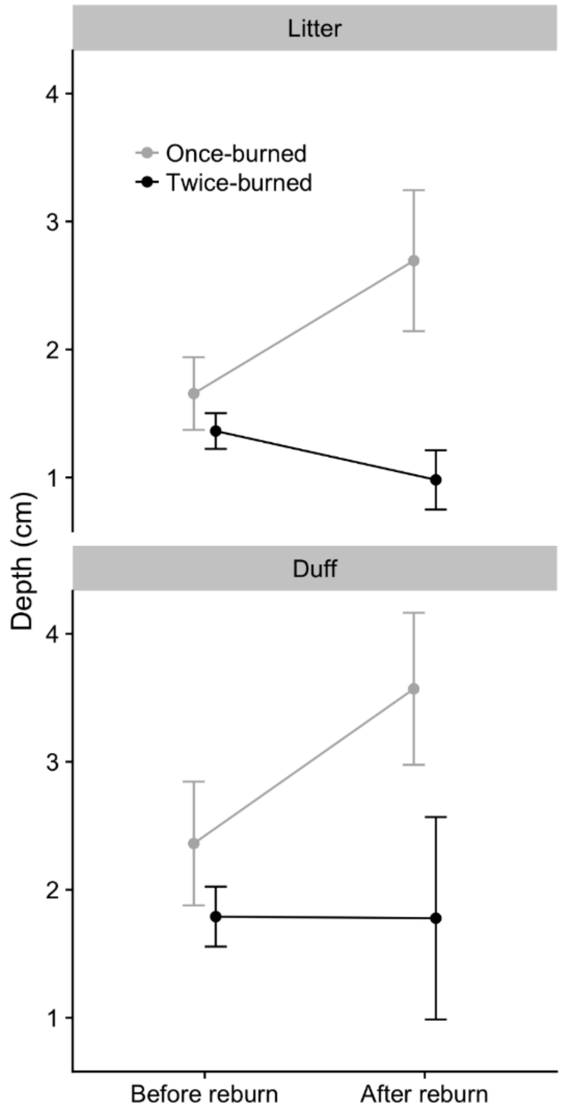

Figure 5. Effects of single and repeat fires on litter and duff fuel depths. Contrasts are identical to those described in Figure 4. Litter is organic material that is finer than 1 hr fuels, but is undecomposed. Duff is partially decomposed organic material. Values are means with vertical bars representing ±1 standard error.

The scope of inference for this BACI study 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 in 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. 2015). 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 (Stevens-Rumman et al. 2016; Coppelatta et al. 2016).

Black carbon study

Twice-burned sites had approximately double the amount of black C on CWD that was present in once-burned sites (Table 3), a statistically significant difference ($P = 0.004$, $W = 87$). Relative variability (coefficient of variation) of black C was about three times higher in once-burned sites (CV = 133) that in twice-burned sites (CV = 41), with both the lowest and highest black C stocks measured in once-burned forests.

Total CWD biomass was significantly greater in once-burned sites compared to twice-burned sites ($P < 0.001$, $W = 3$). Once-burned sites had about double the CWD biomass present in twice-burned sites (Table 3). Relative variability of CWD biomass was similar in once-burned (CV = 32) and twice-burned (CV = 41) sites. In contrast, relative variability of CWD biomass was markedly lower than that of black C biomass in once-burned sites (Table 3). Black C accounted for 0.7% of CWD biomass in once-burned sites and 2.9% of CWD biomass in twice-burned sites.

The finding of greater CWD biomass in once-burned plots compared to twice-burned plots in this study differs from what we found in the BACI plots, likely for two reasons. First, the BACI plots were located in old-growth western larch/mixed-conifer forests, and thus included much larger individual CWD pieces that were not completely consumed in the reburn. Second, the BACI plots sampled the entire range of initial fire severities, from very low to very high (Belote et al. 2015), while the black carbon study deliberately sampled in patches that burned at high severity in the initial fire.

Table 3. Black (pyrogenic) carbon produced on CWD and total CWD biomass in once- and twice-burned mixed-conifer forest in the Bob Marshall Wilderness, Montana, USA.

Black C (kg ha^{-1})	Once-burned n = 10 sites	Twice-burned n = 10 sites
Mean	323	655
Median	239	604
CV	133	41
Range (min-max)	0-1489	285-1133
Total CWD biomass (kg ha^{-1})		
Mean	45894	22755
Median	40956	21761
CV	32	41
Range (min-max)	26116-75823	6001-34787

UOB study: surface and canopy fuels
Fire history (unburned, once-burned, reburned) had complex relationships with woody surface fuel loads (Figure 6). Once-burned plots had either significantly more ($P<0.05$; 1 h, 100 h) or the same amount ($P >0.05$; 10 h, 1000 h) of woody surface fuels compared to unburned plots (Table 4). Reburns, however, reduced woody surface fuels relative to once-burned plots in all but the largest (≥1000 h) size class. The coarse fuel (≥1000 h) did not differ among fire history type. Single and repeat fires in mixed-conifer forests in central Idaho had similar effects on coarse woody debris (Stevens-Rumann and Morgan 2016). There, single high severity fires had similar CWD loads to high-on-low severity and low-on-high severity reburns (Stevens-Rumann and Morgan 2016). Our results are also consistent with the general trends observed for the BACI dataset (Figure 4). Broadly speaking, the primary medium-term effect of an initial fire (especially moderate and high severity fires) after a long fire-free period is to recruit new woody surface fuels as fire-killed trees and branches fall to the forest floor (Dunn and Bailey 2012, 2015). As subsequent fires burn fuels are both consumed and deposited, as they move from the canopy to surface pools, maintaining highly variable surface fuel loads in active fire regime landscapes (Collins et al. 2016, Stevens-Rumann and Morgan 2016).

In contrast, litter and duff in once-burned and reburned plots were significantly lower than in unburned plots, and each was lower in reburned plots than in once-burn plots (Fig 6, Table 4). These fuel classes, especially duff, are not buffered in the same way by transfer of fire-killed material from the canopy to the surface fuel pools.

Shrub fuel loads were lower in both reburn and onceburn plots compared to unburned plots, but were not different from each other (Figure 6, Table 4). The results for shrubs differ somewhat from other regions, especially the Sierra Nevada and southern Cascades where initial high severity fires can promote rapid development by montane chaparral shrub species (Lauvaux et al. 2016), which then establish an amplifying feedback promoting higher reburn severity (Coppoletta et al. 2016). This shrub-mediated pathway is not evident in our data, however. We detected no differences among fire histories of herb fuels (Figure 6, Table 4).

Mean canopy fuel loads declined progressively from unburned, to once-burned to reburned plots, with the steepest declines in the finest fuel classes (Figure 7). Pairwise differences between different fire histories were always significant for all six canopy fuel metrics analyzed (Table 4). Vertical profiles (Figure 8) of mean available canopy fuel load (ACFL) illustrated the progressive reduction of canopy fuels due to the combined effects of fuel consumption during fires and post-fire mortality and transfer of aerial fuels to the surface fuel bed.

Total aboveground biomass in reburned plots was significantly lower than in once-burned and unburned plots (Figure 9, Table 4). There was marginally significant difference between unburned and once-burned plots. Total aboveground biomass was calculated as the sum of all surface fuels (including herbs, shrubs, and the forest floor) plus canopy fuels and tree boles.

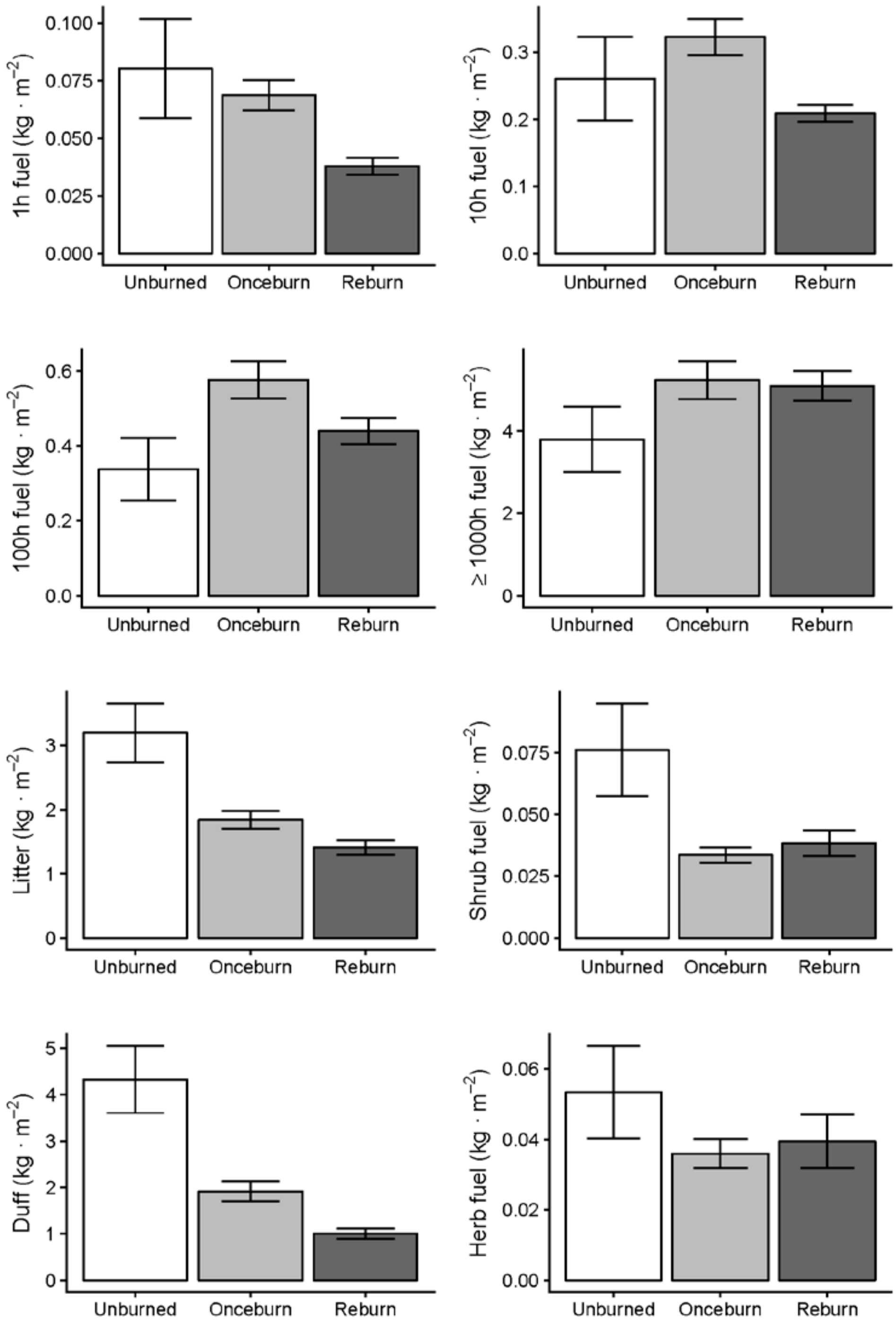

Figure 6. Mean surface fuel loads in unburned (white), once-burned (light grey), and reburned (dark grey) plots in mixed-conifer forest in the Bob Marshall Wilderness, Montana, USA. Vertical lines represent ± 1 standard error of the mean.

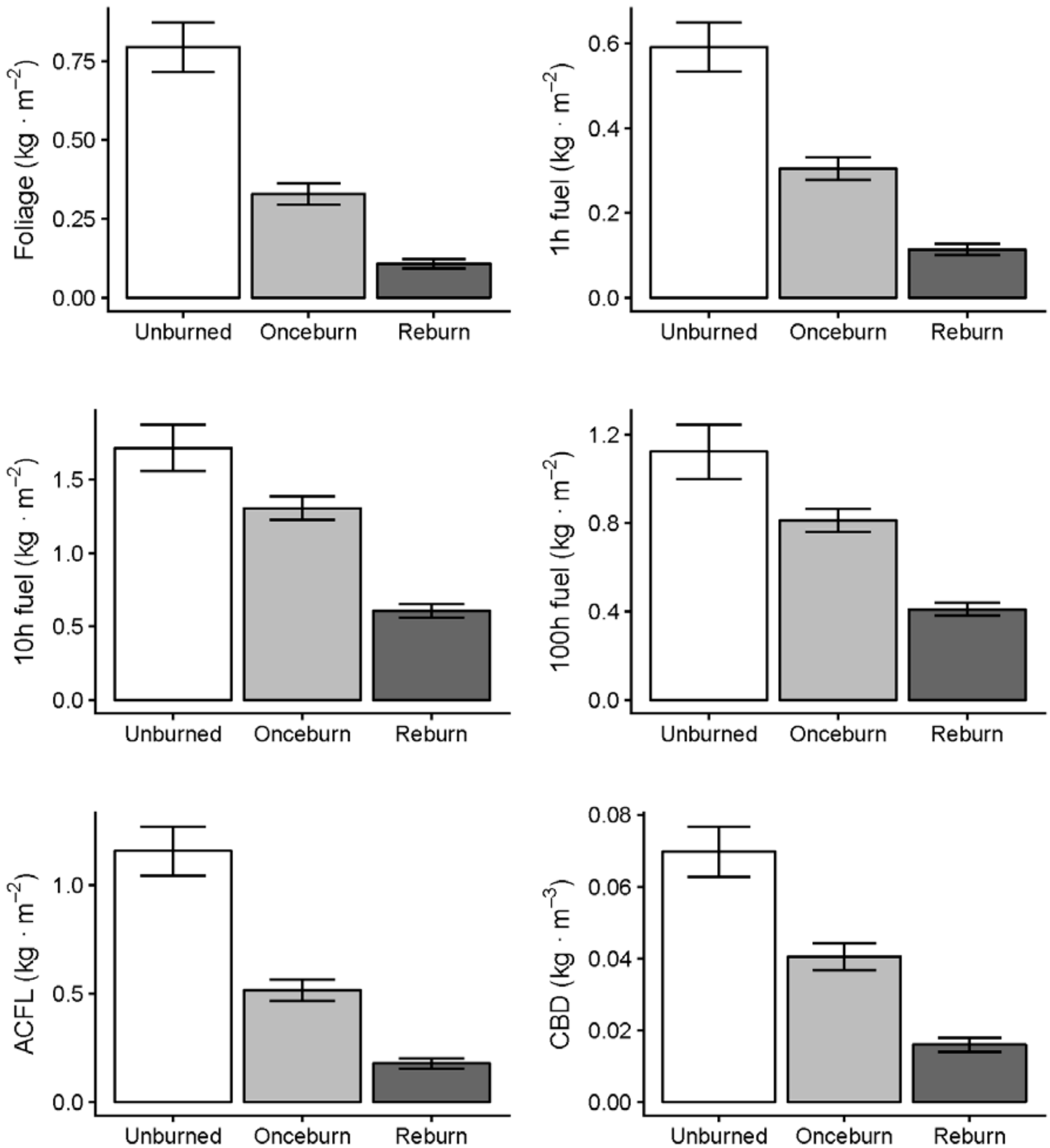

Figure 7. Mean canopy fuel loads in unburned (white), once-burned (light grey), and reburned (dark grey) plots in mixed-conifer forest in the Bob Marshall Wilderness, Montana, USA. 1h – 100h fuels largely consist of branch biomass and smaller stems. ACFL: available canopy fuel load. CBD: canopy bulk density. Vertical lines represent ± 1 standard error of the mean.

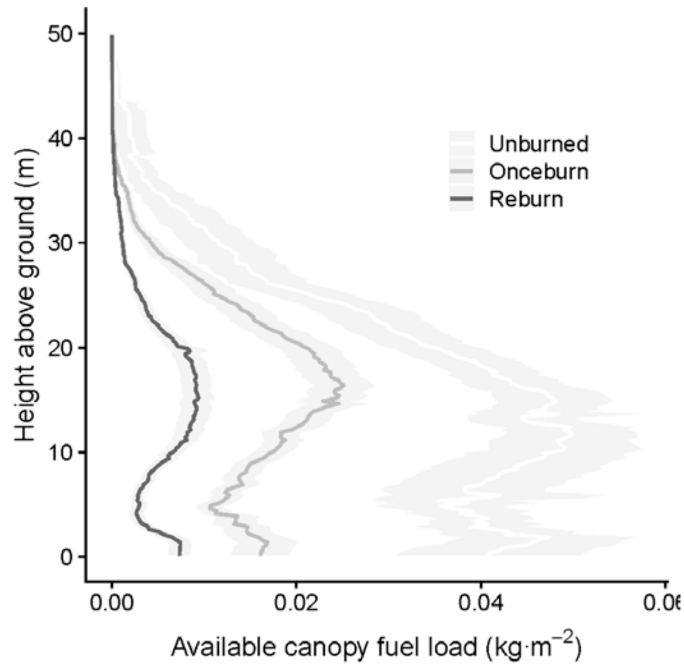

Figure 8. Mean available canopy fuel load (ACFL) profile at 0.25m height increments for unburned (white), once-burned (light grey), and reburned (dark grey) plots in mixed-conifer forest in the Bob Marshall Wilderness, Montana, USA. The ACFL profile incorporates fine fuels from all trees, saplings and seedlings. Bands around lines represent ± 1 standard error of the mean at each height increment.

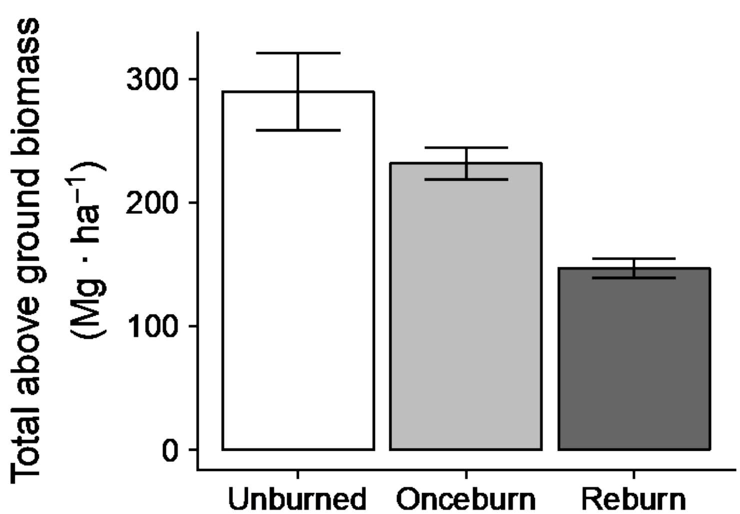

Figure 9. Mean total above ground biomass – all surface and canopy fuels combined – for unburned (white), onceburn (light grey), and reburn (dark grey) plots in mixed-conifer forest in the Bob Marshall Wilderness, Montana, USA.. Vertical lines represent ± 1 standard error of the mean.

Table 4. Results of permutation tests on differences in surface and canopy fuels between unburned, once-burned, and reburned plots. Differences are shown in the respective units of each response variable. Significant contrasts are highlighted in bold font.

Variable	Contrast	Difference	Units	P-value
1h surface fuels	**Onceburn-Reburn**	**0.03**	**kg/m^2**	**< 0.001**
	Unburned-Onceburn	0.01	kg/m^2	0.465
	Unburned-Reburn	**0.04**	**kg/m^2**	**0.007**
10h surface fuels	**Onceburn-Reburn**	**0.11**	**kg/m^2**	**< 0.001**
	Unburned-Onceburn	-0.06	kg/m^2	0.273
	Unburned-Reburn	0.05	kg/m^2	0.360
100h surface fuels	**Onceburn-Reburn**	**0.14**	**kg/m^2**	**0.021**
	Unburned-Onceburn	**-0.24**	**kg/m^2**	**0.042**
	Unburned-Reburn	-0.10	kg/m^2	0.385
≥1000h surface fuels	Onceburn-Reburn	0.14	kg/m^2	0.807
	Unburned-Onceburn	-1.43	kg/m^2	0.203
	Unburned-Reburn	-1.29	kg/m^2	0.241
Litter surface fuels	**Onceburn-Reburn**	**0.43**	**kg/m^2**	**0.026**
	Unburned-Onceburn	**1.35**	**kg/m^2**	**0.001**
	Unburned-Reburn	**1.78**	**kg/m^2**	**< 0.001**
Duff surface fuels	**Onceburn-Reburn**	**0.92**	**kg/m^2**	**0.001**
	Unburned-Onceburn	**2.41**	**kg/m^2**	**< 0.001**
	Unburned-Reburn	**3.33**	**kg/m^2**	**< 0.001**
Shrub surface fuels	Onceburn-Reburn	0.00	kg/m^2	0.509
	Unburned-Onceburn	**0.04**	**kg/m^2**	**0.006**
	Unburned-Reburn	**0.04**	**kg/m^2**	**0.011**
Herb surface fuels	Onceburn-Reburn	0.00	kg/m^2	0.791
	Unburned-Onceburn	0.02	kg/m^2	0.215
	Unburned-Reburn	0.01	kg/m^2	0.300
Foliage	**Onceburn-Reburn**	**0.22**	**kg/m^2**	**< 0.001**
	Unburned-Onceburn	**0.47**	**kg/m^2**	**< 0.001**
	Unburned-Reburn	**0.69**	**kg/m^2**	**< 0.001**
1h canopy fuels	**Onceburn-Reburn**	**0.19**	**kg/m^2**	**< 0.001**
	Unburned-Onceburn	**0.29**	**kg/m^2**	**< 0.001**
	Unburned-Reburn	**0.48**	**kg/m^2**	**< 0.001**
10h canopy fuels	**Onceburn-Reburn**	**0.70**	**kg/m^2**	**< 0.001**
	Unburned-Onceburn	**0.41**	**kg/m^2**	**0.046**
	Unburned-Reburn	**1.11**	**kg/m^2**	**< 0.001**
100h canopy fuels	**Onceburn-Reburn**	**0.40**	**kg/m^2**	**< 0.001**
	Unburned-Onceburn	**0.31**	**kg/m^2**	**0.016**
	Unburned-Reburn	**0.71**	**kg/m^2**	**< 0.001**
ACFL	**Onceburn-Reburn**	**0.34**	**kg/m^2**	**< 0.001**
	Unburned-Onceburn	**0.64**	**kg/m^2**	**< 0.001**
	Unburned-Reburn	**0.98**	**kg/m^2**	**< 0.001**
CBD	**Onceburn-Reburn**	**0.02**	**kg/m^3**	**< 0.001**
	Unburned-Onceburn	**0.03**	**kg/m^3**	**0.002**
	Unburned-Reburn	**0.05**	**kg/m^3**	**< 0.001**

UOR study: forest structure and live tree density

We detected no overall differences in stem density (where trees are defined as stems ≥1.37 m tall) among fire histories (Table 5). While stem densities were not significantly different due to high variability, general relationships between stem density and fire history did follow expectations, with once-burned and reburned sites having higher average tree densities unburned sites due to high densities of post-fire tree regeneration. Basal area differed strongly and significantly between both once-burned and reburned sites, and reburned and unburned sites (Table 5). Burned plots – especially reburn plots – had many more stems in smaller DBH classes than unburned plots, but fewer stems in larger DBH classes (Figure 10). This accounts for observed differences in basal area but not in stem density. Our basal area results agree well with those of Stevens-Rumann and Morgan (2016) who found that basal area was generally lower in reburned sites compared to once-burned sites, with the exception of low severity fires compared to low severity reburns of an initial low-severity fire, which were not significantly different and had much higher basal areas than other fire severity × history combinations in their central Idaho study area.

Diameter distributions of living and dead stems were strikingly different among the three fire histories (Figure 10). Unburned plots were characterized by a rotated-sigmoid diameter distribution, while the once-burned and reburned plots exhibited the reverse-J shaped distributions. Fire effects were evident in the reduced abundance of trees in medium and large DBH size classes on burned sites, with fewer live stems in reburned sites compared to once-burned sites (Figure 10). These general trends mirror patterns of canopy fuels across the fire history gradient from unburned to reburned (Figures 7 and 8).

Table 5. Results of permutation tests on differences in forest structure variables in unburned, once-burned, and reburned plots. Differences are shown in the respective units of each response variable. Significant contrasts are highlighted in bold font.

Variable	Contrast	Difference	Units	P-value
Basal Area	**Onceburn-Reburn**	**11.64**	**ba/ha**	**< 0.001**
	Unburned-Onceburn	3.51	ba/ha	0.387
	Unburned-Reburn	**15.15**	**ba/ha**	**< 0.001**
Stem density	Onceburn-Reburn	642.62	n/ha	0.367
	Unburned-Onceburn	-1780.88	n/ha	0.169
	Unburned-Reburn	-1138.26	n/ha	0.415
Total above ground biomass	**Onceburn-Reburn**	**84.89**	**Mg/ha**	**< 0.001**
	Unburned-Onceburn	58.18	Mg/ha	0.065
	Unburned-Reburn	**143.07**	**Mg/ha**	**< 0.001**

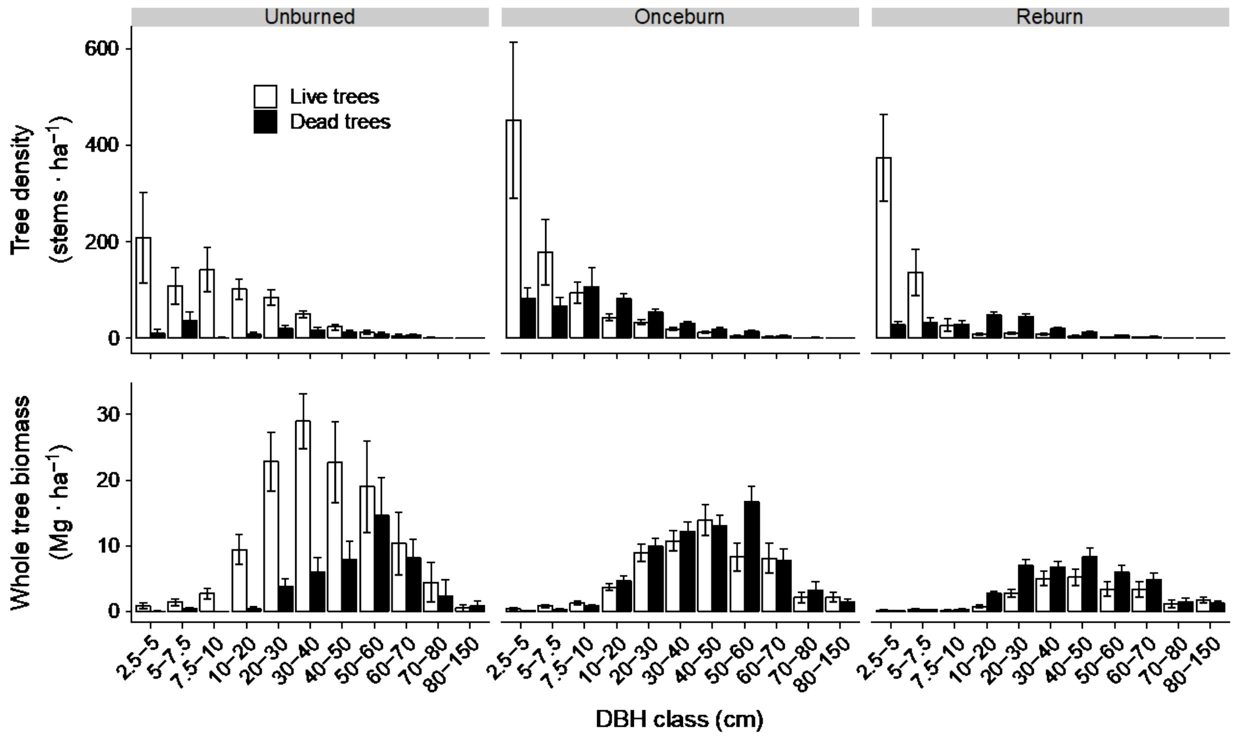

Figure 10. Mean tree density and whole tree biomass distributions by diameter at breast height (DBH) class for unburned, once-burned, and reburned plots (left to right, respectively) in mixed-conifer forest in the Bob Marshall Wilderness, Montana, USA. Live tree contributions are represented by white bars, dead trees by black bars. Vertical lines represent ± 1 standard error of the mean.

Total live tree density was unrelated to any fire or environmental variables, except for topographic aspect. Areas with more southerly-facing slopes tended to have fewer live trees. However, live tree density was strongly predicted by an interaction between slope and initial fire severity. Steeper slopes that burned with higher fire severity tended to have fewer live trees, than gentler slopes burning under higher severity (Figure 11). We also observed an interaction between slope and aspect where steeper slopes with more southerly aspects had fewer live trees compared to steeper slopes with more northerly aspects (Figure 12). These results suggest a predictable, bottom-up control of topographic setting on forest density. Steep south aspects—relatively hot, dry environments—mediate long-term fire effects on forest density by slowing tree regeneration and recruitment.

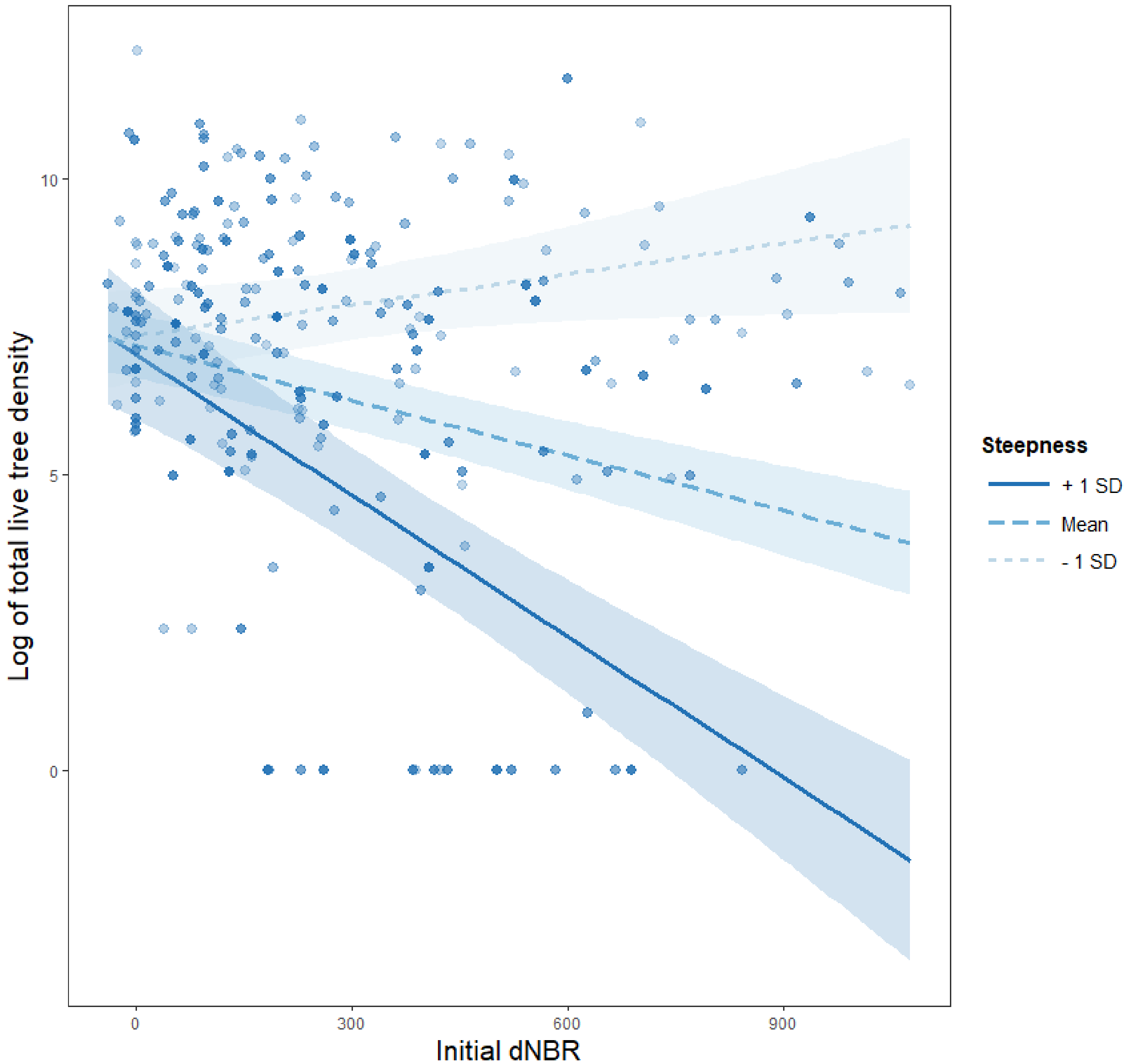

Figure 11. Interactive effects of initial fire severity (dNBR) and steepness of topography on live tree densities. The effects of fire severity depended on the steepness of slope. Steeper slopes that burned more severe had fewer live trees compared to similar initial fire severity on more gentle slopes.

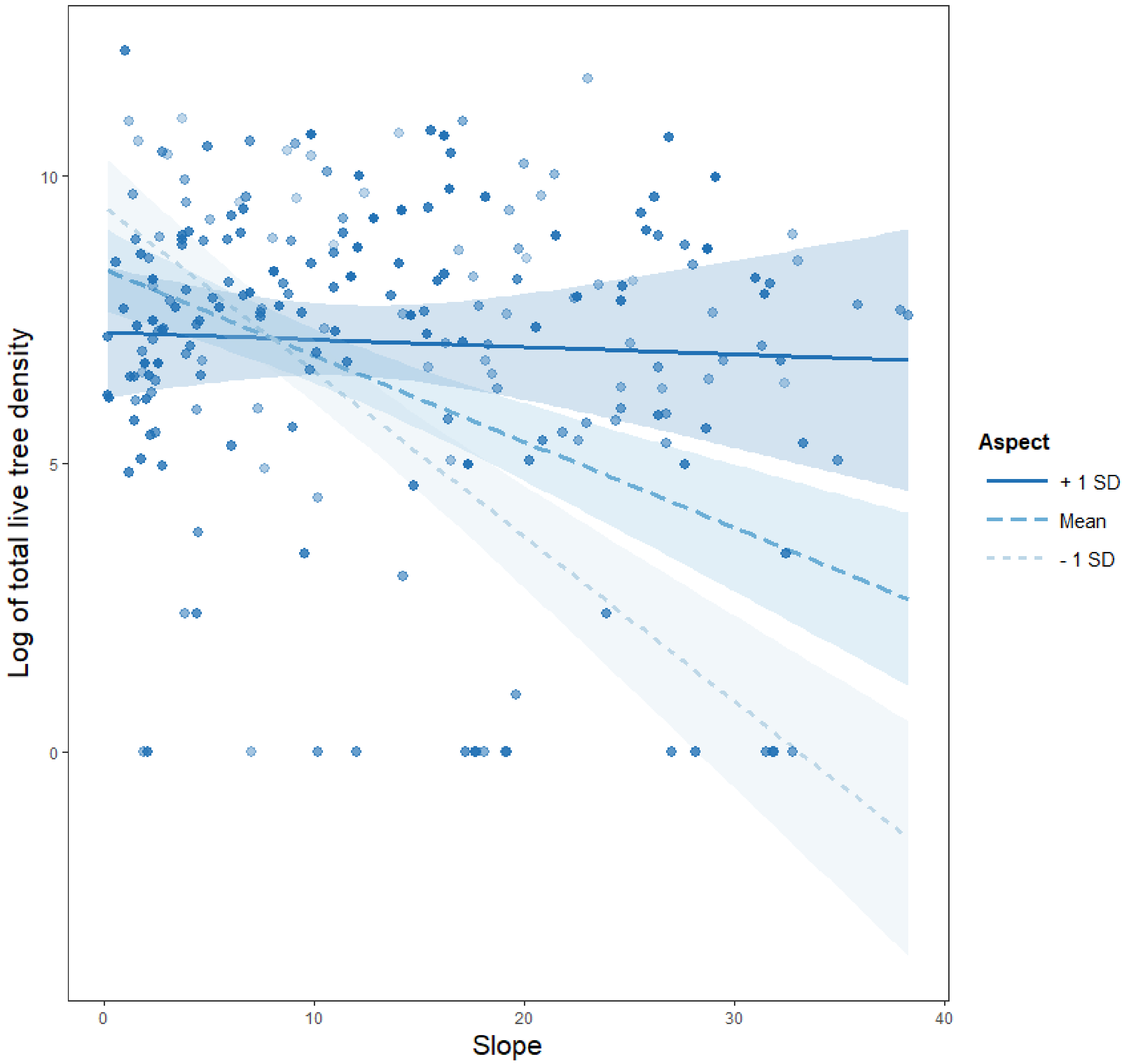

Figure 12. Slope and aspect interact to influence live tree density. Steep slopes with more southerly aspects tended to be characterized by lower tree densities compared to slopes with more northerly aspects.

UOR study: fire severity and state shifts to non-forest

Seventeen (of 224) plots were classified as "non-forest" based on our simple ruleset (i.e., no live seedlings, saplings, or trees) used to identify putative states. Seven non-forest plots were identified in once-burned plots, and ten plots were identified as non-forest in reburns. As a proportion of total plots sampled, putative shifts to non-forest occurred in similar numbers between once (7.9% shifted to non-forest) and twice (8.3% shifted to non-forest) burned plots. When all plots were included in a random forest model heatload, dNBR of the first fire, and slope emerged as key variables explaining the environmental settings and fire severity where putative state shifts occurred (Figure 13 right panel). When we ran the random forest model using a classification based only on seedling density (plots classified based on no live seedlings), the same three variables emerged as key variables explaining fire and environmental settings without seedings (Figure 14 left panel). When re-running the same analysis for the subset of plots that experiences a reburn, heatload and slope also emerged as important variables discriminating putative states in reburns.

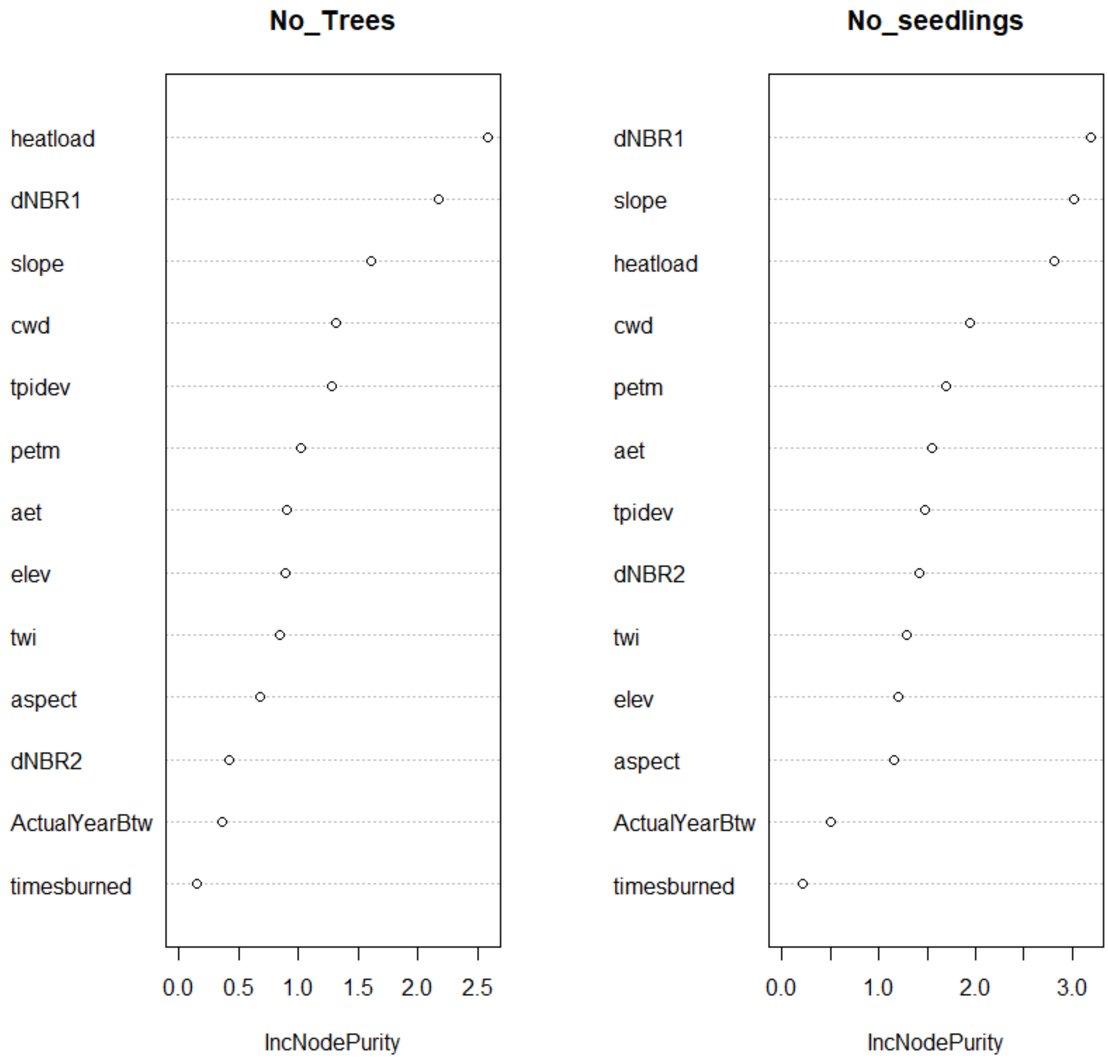

Figure 13. Variable importance scores ("IncNodePurity") from random forest models across all plots ranked from most important to least. Variables with a high score were more frequently included as good predictors in random forest models to determine differences between plots with no living trees (i.e., those that have shifted to a new putative non-forest state) and those plots with some living trees. Left panel includes a classification based on density of all trees, saplings, and seedlings, and the right panel includes plots classified based only on density of live seedlings. Variables are heatload = heat load index; dNBR1 = delta normalized burn ratio from the 1st fir; slope = slope angle; cwd = climate water deficit; tpidev = normalized topographic position index; petm = potential evapotranspiration; aet = actual evapotranspiration; elev = elevation; twi = topographic wetness index; aspect = slope aspect; dNBR2 = dNBR of the second fire; ActualYearBtw = years between reburned plots; timesburned = number of times a plot burned in the last 30 years.

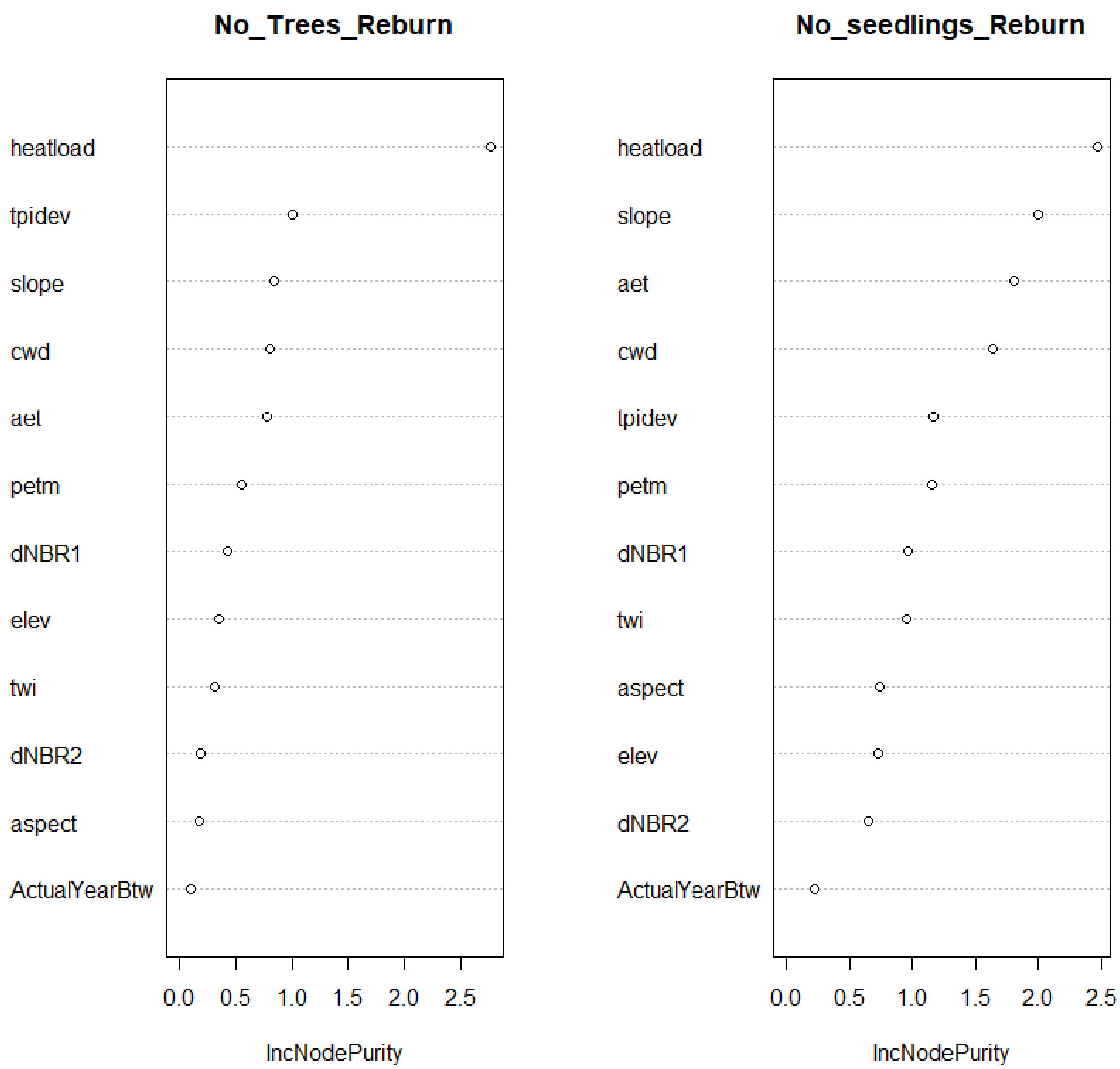

Figure 14. Variable importance scores from random forest models across plots that reburned ranked from most important to least. Variables with a high score were more frequently used as good predictors in random forest models used to determine differences between plots with no living trees (i.e., those that have shifted to a new putative non-forest state) and those plots with some living trees (i.e., those that could potentially maintain some tree cover). Note that for reburned plots, initial or reburn fire severity are not in the top half most important variables in discriminating between plots with or without any tree (left panel) or those with or without any tree seedlings (right panel).

Figure 15. A steep, southwest facing slope that burned in 1985 and reburned in 2003. This site supported no live trees or tree seedlings when sampled in 2016, and illustrates the typical attributes of sites in this study that have transitioned to a putative non-forest state. Note the abundant conifer regeneration directly down slope and on the opposing northeast facing slope, all of which burned in the 1985 and 2003 fires. While this example site reburned, we found that topographic setting (hot, dry southern aspects) and initial fire severity were most important in predicting sites that transitioned to non-forest, not number of times a site burned (Figure 13).

Initial fire severity was related to reburn fire severity, though the relationship was characterized by a high degree of variability. More severe observations of dNBR of the first fire tended to be related to lower severity of reburns (Figure 16). However, severity of the initial fire or reburn did not occur in the top half of variables explaining putative states among reburned plots (Figure 14). Thus, while reburns unambiguously consume surface fuels (Figure 7), and alter canopy fuels and forest structure (Figures 8, 9, 10), they appear to have relatively little influence on transitions to non-forest communities.

The moderating effect of initial fire severity on reburn fire severity (Figure 16) is consistent with the results of Parks et al. (2014), who found that reburned areas had lower mean and median dNBR values than once-burned areas. Harvey et al. (2016) found that, within the perimeter of twice-burned areas through the northern US Rocky Mountains, in mid-montane forests the frequency of high-severity pixels decreased in the second fire, while the frequency of high-severity pixels in the second fire increased slightly in subalpine forests. Both of these remote sensing based studies differ in their methods from our approach. In Figure 16 we show the relationship of dNBR of the first fire to the dNBR of the second fire, for fixed sample points. A major limitation of satellite derived measures of fire severity is their inability to detect changes of overall fuel amount or arrangement, and their inability to detect cumulative fire effects in reburned areas.

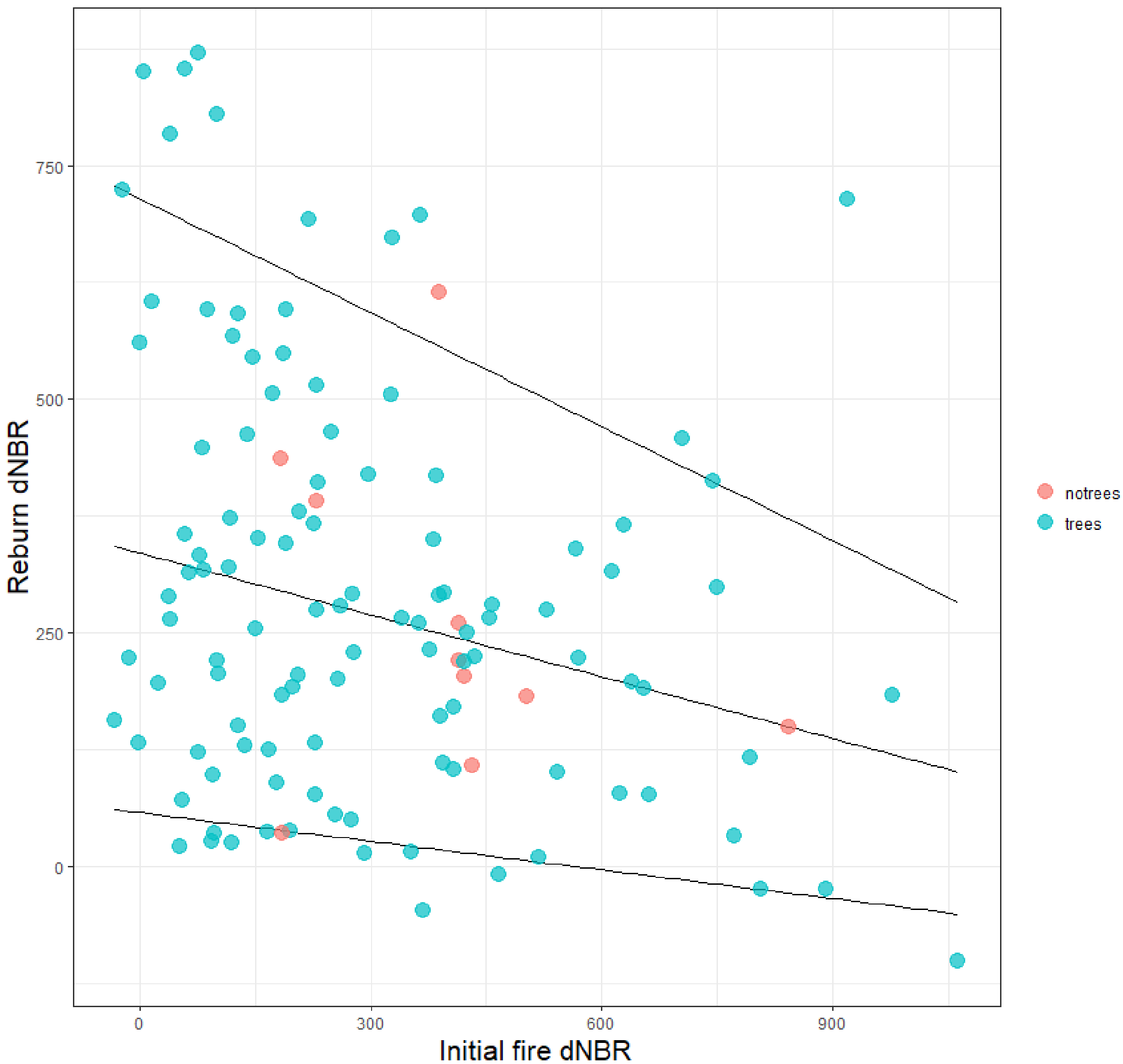

Figure 16. Initial fire severity tended to reduce the severity of reburns. Quantile regression linear models showing fits through the 90th, 50th, and 10th percentiles of the data (top to bottom predicted line, respectively) suggest more severe initial fires burned less severe during a reburn event. Plots are classified as either "notrees" and "trees" based on simple rulesets described in the methods and are shown for comparisons.

Science delivery activities

We maintained a very active and successful science delivery and outreach program during this project. Science delivery methods included traditional scientific publications and presentations, participation in a JFSP Fire Science Exchange Network workshop, public presentations, and multiple interviews with print and digital journalists. This diversified science delivery program reached managers, scientists, students, and the general public through multiple platforms.

Traditional science delivery activities included five contributed oral presentations and one contributed poster presentation delivered at regional, national, and international scientific meetings. Additionally, project PI Larson delivered an invited workshop presentation at the Wilderness Fire Management Workshop held July 14-15, 2016 at the Spotted Bear Ranger Station, Montana. This workshop was co-organized and co-hosted by the JFSP Northern Rockies Fire Science Network and the Bob Marshall Wilderness Complex Managers and Spotted Bear Ranger District, Flathead National Forest. These technical presentations were complemented by an invited public presentation about this project delivered to an audience of approximately 150 at the Bob Marshall Wilderness Foundation 2018 Wilderness Speaker Series on March 7, 2018. This speaker series is attended by interested members of the public and wilderness advocates, as well as regional public land managers. Other science delivery products include one published peer-reviewed journal article, one refereed conference proceeding (currently in press), and two additional peer-reviewed journal articles currently in preparation.

Multiple media contacts and interviews resulted in this research being featured in two different news articles, a book, and a short documentary film. An all-day interview about this research project with author Ed Struzik lead to inclusion of this study (including photos) in the book *Firestorm* (Island Press). This project was also featured in the film, *Wild Science: Wilderness and Fire*, a short documentary film commissioned by the US Forest Service Aldo Leopold Wilderness Research Institute and produced by High Plains Films. A videographer from High Plains Films spent two days filming our field crew as they collected field data in the study area. This field visit was followed with an interview of PI Larson. The film features extensive quotes from Larson, as well as footage of the 2016 field crew sampling in the study area.

Conclusions

We achieved our overarching goals for this study, which were to quantify and compare the effects on forest ecosystem structure of single and repeat wildfires; to determine if past wildfires influence the severity of subsequent wildfires; and to determine if short-interval reburns are causing transitions to non-forest communities.

Analysis of the new field data collected in this study shows the strong effects of single and repeat fires on forest structure and fuels. Surface and canopy fuels exhibited contrasting responses, with the surface fuel complex strongly buffered by post-fire inputs from the overstory. Thus, it is not appropriate to consider an initial fire a fuel treatment—except perhaps for the duff layer (Figure 6). Indeed, for some fuel classes loadings are elevated in the years following an initial fire relative to unburned sites. A second fire is required to cause meaningful reductions to the surface fuel loads, and three or more fires may be required to reduce the largest (≥1000 h) fuels (Figure 6), except in situations where high severity patches are reburned (Table 3). Fire effects on canopy fuels are much more predictable, with steady reductions in ACFL (Figure 8), tree biomass (Figure 10), and total aboveground biomass (Figure 9) from unburned, to once-burned, to reburned sites. Our most important finding with respect to fire effects on fuel loads and forest structure is that surface fuel loads are maintained or increased in the years following an initial wildfire after a long fire-free period as fire killed trees and branches fall to the ground. This unsurprising result nevertheless deserves highlighting because the current conventional wisdom (e.g., Parks et al. 2014) is that an initial fire can be thought of as a "fuel treatment." Our results clearly demonstrate that, while reburns function as fuel treatments, initial fires either maintain or elevate most woody surface fuel pools.

Using a satellite derived measure of fire severity (dNBR), we found that severity of reburns tended to be negatively related to the severity of the initial fire (Figure 16). This is generally consistent with the results of Parks et al. (2014) and Harvey et al. (2016), the two most similar studies to our study area in terms of forest composition and geography. It is also consistent with results from other regions that have shown a moderating influence of initial fires on reburn severity using similar remote sensing methods (Coop et al. 2016, Coppoletta et al. 2016). However, we caution that satellite derived assessments of reburn severity are of limited utility, and potentially misleading. Because satellite based assessments of reburn severity do not consider cumulative effects of both the first and second fires—severity is assessed only with respect to immediate pre-fire conditions, regardless of whether the fire is a first entry after a long fire-free period, or a short-interval reburn—the finding of moderated reburn severity is ecologically misleading. Our field measurements clearly demonstrate the cumulative effects of reburns, which reduce surface fuels, canopy fuels, overstory basal area and biomass, and total aboveground biomass relative to a single fire. Continuing to base assessments of reburn severity and effects on satellite based measures like dNBR will lead to misrepresentation of actual conditions on the ground created by repeated fires. We suggest reburn effects will be best assessed using field measurements or active remote sensing methods optimized to detect different forest structures such as LiDAR, in addition to or instead of traditional satellite based methods such as dNBR.

Our most surprising finding was the relative unimportance of reburning in causing transitions to putative non-forest states. We expected based on ecological theory (Paine et al. 1998) and results from other regions (e.g., Coop et al. 2016) that reburns would be the principal driver of transitions to non-forest conditions. Instead, we found that environmental variables related to topographic position and the severity of the initial fire (Figures 11, 12, and 13), but not necessarily the occurrence of a reburn, were far more important in explaining transitions to non-forest. While reburns magnify the impacts of the initial fire, for example by killing seedlings and further reducing local seed sources, the main factors required to trigger transition to non-forest were the occurrence of an initial high-severity fire on a steep southerly aspect. We expected very severe reburn effects arising from elevated fuel loads caused by an initial high-severity fire to be more important (e.g., as in Coppoletta et al. 2016), and largely independent of topographic position. That hypothesis was not supported by our data. Rather, we see strong bottom up control of transitions to non-forest mediated by topography. This suggests a degree of predictability of such transitions, which could be very informative to managers attempting to design and implement landscape scale forest restoration and climate change adaptation treatments (Hessburg et al. 2015).

Implications for management

Our results show that, if the goal is to use managed wildfires as fuel reduction treatments, managers need to plan for multiple fire entries (i.e., two or more fires) before surface fuels will be significantly reduced. Managers need to anticipate that an initial fire after a long fire-free period will result in similar or even elevated fuel loadings within about a decade after the first fire. This underscores the need for a second entry to reduce fuels, if that is a management objective, before transitioning to a program of maintenance fires. The appropriateness and importance of allowing natural ignitions to burn as managed wildfires is self-evident in cases where managers seek to conserve ecosystem function and manage fires for ecological benefit. Here, we focus on the question of wildfires as fuel reduction treatments, finding that two successive fires are required for meaningful reduction of surface fuels loads.

Our results demonstrate that some transitions to a putative non-forest condition is to be expected following both initial fires and short-interval reburns. Thus, managers may wish to incorporate this outcome into their expectations, and especially outreach and education efforts, in order to prepare policy makers and the public. This is not necessarily bad, as historical reconstructions have shown that many formerly fire-maintained open areas have been encroached by forest during the period of fire exclusion (Hessburg et al. 2015)—returning some areas to a putative non-forest condition may actually be restorative from a landscape perspective. Given climate change and increasingly hot, dry summers in western Montana, fire-triggered transitions to non-forests are likely to increase in frequency and area in coming decades.

Finally, our analyses suggest that a broader range of post-treatment conditions than the structurally complex conditions 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 (Figure 17).

Figure 17. (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).

Future research

Large, active fire regime landscapes like the Bob Marshall Wilderness offer value to society as reference ecosystems that provide a source of scientific information to managers seeking to design and implement forest landscape restoration and climate change adaptation treatments. The next applied fire science and management research that follows from this work is to magnify plot based field studies and satellite-based fire severity studies with high-resolution, large spatial extent, forest structure data obtained from LiDAR. Such data will enable scientists and managers evaluate, and apply to land management, linkages between topography, fire history, and forest structure and fuels, in a spatially explicitly framework.

Another significant discovery from our field data is the apparent great resilience of northern Rocky Mountain mixed-conifer forests. Despite the dominance of mixed and high-severity fire regimes in this region, fire-mediated transitions to non-forest appear much less widespread than in other regions, such as in forests of the Southwest and Sierra Nevada. We suspect that this is due to multiple factors, including the relatively cooler, wetter climate of our study area compared to more southern locations (Coop et al. 2016); the more important role of highly flammable and fast growing shrubs, and their contribution to high-severity reburns (Coppoletta et al. 2016); and potentially differences in tree species assemblages and the relatively frequency of key traits, such as serotiny and early reproductive maturity of lodgepole pine, which appear to confer forest resilience to short-interval reburns in the northern Rockies. Identifying mechanisms that mediate transitions to non-forest communities, and their commonalities and differences across forest regions, is an important line of future research, especially considering the expected effects of a warmer, drier climate in the coming decades. Such information is needed to develop an overall management framework that uses prescribed fire and managed wildfire to effect adaptive landscape resilience.

Figure 18. Landscape mosaic of the central portion of the study, near the confluence of the White and South Fork Flathead rivers (left background). This view shows the complex mosaic of forest structures created by multiple burns and reburns of variable severity that occurred between 1985 and 2013. Such active fire regime forest landscapes have untapped value as reference ecosystems to guide planning and implementation of large scale forest landscape restoration and climate change adaptation efforts.

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Appendix A: Contact Information for Key Project Personnel

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Appendix B**: **List of Completed/Planned Scientific/Technical Publications/Science Delivery Products:

1. Articles in peer-reviewed journals

Ward, A., C.A. Cansler, and A.J. Larson. 2017. Black carbon on coarse woody debris in once- and twice-burned mixed-conifer forest. Fire Ecology 13(2): 143-147.

Belote, R.T., C.T. Maher, and A.J. Larson. Topography mediates shifts to non-forest communities following single and repeat fires in mixed-conifer forest. In preparation.

Larson, A.J., C.T. Maher, and R.T. Belote. Do wildfires function as fuel treatments? Fuels and forest structure in unburned, once-burned, and twice-burned mixed-conifer forest. In preparation.

2. Technical reports

None.

3. Text books or book chapters

None.

4. Graduate thesis

None.

5. Conference or symposium proceedings scientifically recognized and referenced

Larson, A.J., J.K. Berkey, C.T. Maher, W. Trull, R.T. Belote, and C. Miller. Fire history (1889-2017) in the South Fork Flathead River watershed within the Bob Marshall Wilderness, including effects of single and repeat wildfires on forest structure and fuels. *In press.* In: Proceedings of the Fire Continuum Conference. USDA Forest Service General Technical Report RMRS-GTR-XXXX.

6. Conference or symposium abstracts

Larson, A.J., J.K. Berkey, C.T. Maher, W. Trull, R.T. Belote, and C. Miller. Fire history (1889-2017) in the South Fork Flathead River watershed within the Bob Marshall Wilderness, including effects of single and repeat wildfires on forest structure and fuels. Fire Continuum Conference. May 21-24, 2018. Missoula, Montana, USA. Contributed oral presentation.

Larson, A.J. and R.T. Belote. Effects of single and repeat fires on forest structure and fuel loads in mixed-conifer forest. 7th International Fire Ecology and Management Congress. November 28-December 2, 2017. Orlando, Florida, USA. Contributed oral presentation.

Larson A.J. and R.T. Belote. Forest structure and fuel loads following single and repeat fires in mixed-conifer forest. 11th North American Forest Ecology Workshop. June 19-22, 2017. Edmonton, Alberta, Canada. Contributed oral presentation.

Trull, W. and A.J. Larson. Comparing changes in fuel loading, tree regeneration, and forest structure in once- and twice-burned mixed-conifer forests. University of Montana Conference on Undergraduate Research. April 15, 2016. Missoula, Montana, USA. Contributed oral presentation.

Larson, A.J., and R.T. Belote. Spatial heterogeneity and structural development pathways following fire in western larch/mixed-conifer forest. 9th International Association of Landscape Ecology World Congress. July 5-10, 2015. Portland, Oregon, USA. Contributed oral presentation.

7. Posters

Ward, A., A.J. Larson, and C.A. Cansler. Charcoal production in mixed-conifer forest under high severity initial fire and repeat burns. Northwest Scientific Association 86th Annual Meeting. April 1-4, 2015. Columbia Basin College, Pasco, Washington, USA. Contributed poster presentation.

8. Workshop materials and outcome reports

Larson, A.J. 2016. Restoring forest structure and fire regimes in the Bob Marshall Wilderness Complex: lessons from wilderness fire. Invited workshop presentation at the Wilderness Fire Management Workshop. July 15, 2016. Spotted Bear Ranger Station, Montana. This workshop was co-organized and co-hosted by the JFSP Northern Rockies Fire Science Network and the Bob Marshall Wilderness Complex Managers and Spotted Bear Ranger District, Flathead National Forest. https://www.nrfirescience.org/event/wilderness-fire-workshop-and-field-trips

9. Field demonstration/tour summaries
None.

10. Website development
None.

11. Presentations/webinars/other outreach/science delivery materials

Project PI Larson was interviewed about this project by author Ed Struzik, June 12, 2018. Project was featured (including photos) in resulting article published in Yale Environment 360. https://e360.yale.edu/features/does-a-fire-ravaged-forest-need-human-help-to-recover

Larson, A.J. 2018. The untrammeled observatory: lessons from wilderness fire. Bob Marshall Wilderness Foundation 2018 Wilderness Speaker Series. March 7, 2018. Flathead Valley Community College, Kalispell, Montana, USA. Invited oral presentation.

Project PI Larson was interviewed twice by reporter Matt Blois about this project, leading to news feature (including photos) on wilderness fire management in the forestry news outlet *Treesource*. “Wilderness areas are a laboratory for fire scientists, managers.” https://treesource.org/news/management-and-policy/wilderness-areas-are-a-laboratory-for-fire-scientists-managers/

Wild Science: Wilderness and Fire. Documentary film commissioned by the Aldo Leopold

Wilderness Research Institute and produced by High Plains Films. The film features extensive quotes from project PI Larson, as well as footage of the 2016 field crew sampling in the study area. https://www.youtube.com/watch?v=lqzIiwvoJVk&t=18s

Larson, A.J. 2016. Film interview about this research with Dru Carr of High Plains Films for documentary film on wilderness fire commissioned by the Aldo Leopold Wilderness Research Institute. August 15, 2016. Missoula, Montana.

High Plains Films spent two days filming the crew conducting field data collection in the study area. Footage of our field crew was included in a documentary film on wilderness fire commissioned by the Aldo Leopold Wilderness Research Institute. July 21-22, 2016.

Larson, A.J. 2016. All day interview about this research project with author Ed Struzik leading to inclusion of this study (including photos) in the book *Firestorm* (Island Press).

Appendix C: Metadata

Dataset 1. Submitted to USFS Research Data Archive

Citation_Information:
Originator: Larson, Andrew J.
Originator: Belote, R. Travis
Originator: Berkey, Julia K.
Originator: Maher, Colin T.
Publication_Date: 2018
Title:
Effects of single and repeat wildfires on forest structure and fuels in the South Fork Flathead watershed within the Bob Marshall Wilderness
Geospatial_Data_Presentation_Form: tabular digital data
Publication_Information:
Publication_Place: Fort Collins, CO
Publisher: Forest Service Research Data Archive
Online_Linkage: WILL ASSIGN
Description:
Abstract:
Wilderness areas offer value to society as a source of scientific information. In 2011 and 2012 we collected data on tree stands, seedling and sapling regeneration, fuel loads, and ground cover across 30 sites within South Fork Flathead River Valley of the Bob Marshall Wilderness. All 30 sites had burned within the last decade, which allowed for investigation into the drivers of fire effects that produce heterogeneous post-fire tree and stand-level mortality. Tree survival 8–13 years after fire depended on complex interactions between species, size, and initial burn severity. Following the initial round of data collection, 10 sites reburned. These sites were directly across the Flathead River from 10 sites that remained once-burned, which allowed us to quantify the effects of a recent reburn on forest structure and fuels using a before-after-control-impact study design. Data included covers tree stand data, regeneration data, fuels, and estimates of ground cover. Managers can use this data set to inform the design and monitoring of forest landscape restoration prescriptions.
Purpose:
To quantify the effects of an initial fire and subsequent reburn on forest structure and fuels, as well as the drivers of tree mortality on an individual tree and community level.

Dataset 2. Submitted to USFS Research Data Archive

Citation_Information:
Originator: Larson, Andrew J.
Originator: Belote, R. Travis
Originator: Maher, Colin T.
Originator: Berkey, Julia K.
Publication_Date: 2018
Title:
Forest structure, regeneration, and fuels in unburned, once-burned and twice-burned mixed-conifer forests
Geospatial_Data_Presentation_Form: tabular digital data
Publication_Information:
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Publisher: Forest Service Research Data Archive
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Description:
Abstract: Since the middle 1980s, managers have allowed many naturally ignited wildfires to burn with minimal interference in the Bob Marshall Wilderness, Montana, USA. This contemporary active fire regime has produced a mosaic of recent burn histories comprising various combinations of fire frequency and fire severity, the effects of which are not confounded by past management (e.g., timber harvest) or suppression.

Our study area was the valley floor and lower sidewalls of the main South Fork Flathead River valley and major tributaries between 1233 and 1740 m above sea level. We used a stratified random sampling design to ensure adequate sampling of topographic and fire severity gradients, which we hypothesized would influence post-fire fuel loads, forest structure, and tree regeneration. We sampled n = 224 plot locations distributed among long-unburned (n = 15), once-burned (n = 89), and twice-burned (n = 120) fire histories. Woody surface fuels were sampled using the planer intercept method; herbs, graminoids and shrubs were sampled using Keane's photoload technique; and forest structure, composition, and canopy fuels with sampled with tree measurements in concentric, fixed-area plots. Field sampling occurred during the summers of 2015, 2016, and 2017. Plot-level data include treatment type (unburned, once-burned, twice-burned), year of fire(s), fire severity (dNBR), and date of sampling, plot locations (UTM), in addition to field-measured fuels and forest structure data.
Purpose: The purpose of this study was to investigate the comparative effects of single and repeat wildfires occurring between 1985 and 2013 on surface fuels, canopy fuels, forest structure, and tree regeneration, asking if wildfires function as fuel reduction treatments. We also sought to determine if and under what conditions short-interval reburns cause transitions to a putative non-forest state.