



ORIGINAL RESEARCH

Open Access



Fuel dynamics after a bark beetle outbreak impacts experimental fuel treatments

Justin S. Crotteau^{1,2*}, Christopher R. Keyes¹, Sharon M. Hood³, David L. R. Affleck¹ and Anna Sala⁴

Abstract

Background: Fuel reduction treatments have been widely implemented across the western US in recent decades for both fire protection and restoration. Although research has demonstrated that combined thinning and burning effectively reduces crown fire potential in the few years immediately following treatment, little research has identified effectiveness of thinning and burning treatments beyond a decade. Furthermore, it is unclear how post-treatment disturbances such as a bark beetle outbreak affect fuel treatment effectiveness.

Results: We evaluated differences in surface and canopy fuel characteristics and potential fire behavior metrics between fuel reduction treatments (no-action or control, burn-only, thin-only, thin+burn) implemented in ponderosa pine (*Pinus ponderosa* Lawson & C. Lawson)—Douglas-fir (*Pseudotsuga menziesii* [Mirb.] Franco)-dominated forests that were subsequently affected by a mountain pine beetle (*Dendroctonus ponderosae* Hopkins) outbreak after treatment. Experimental units were measured in 2002 (immediately following fuel treatment) and in 2016 (14 years after treatment and at least 4 years following a beetle outbreak). We found that beetle-altered thinning treatments (thin-only and thin+burn combined) had less fuel (*i.e.*, 34% and 83% lower fine and coarse woody debris loading, respectively) and lower crown fire potential (*i.e.*, 47% lower probability of torching and 42% greater crowning index) than corresponding unthinned treatments (control and burn-only). There was no post-beetle-outbreak effect of burning treatments (burn-only and thin+burn combined) on surface fuel loading, but burning reduced crown fire potential (*i.e.*, 37% greater crowning index) over unburned units (control and thin-only) 14 years after treatment. Additionally, we determined the relative impacts of fuel treatments and the bark beetle outbreak on fuel and crown fire potential differences and found that bark beetle-caused tree mortality inflated differences between controls and thinned treatments (thin-only and thin+burn) for surface fuel loading and probability of torching, but diminished differences between these treatments for canopy fuel loading, canopy bulk density, and crowning index.

Conclusions: Despite the differential effects of bark beetle-caused tree mortality in the treatments, our study suggests that the effects of fuel treatments on mitigating crown fire potential persist even after a stand-transforming insect outbreak, especially when thinning and burning are combined.

Keywords: *Dendroctonus ponderosae*, disturbance interaction, fire and fire surrogate study, fuel dynamics, mediation analysis, *Pinus ponderosa*, *Pseudotsuga menziesii*, treatment longevity

* Correspondence: jcrotteau@fs.fed.us

¹University of Montana, WA Franke College of Forestry and Conservation, 32 Campus Drive, Missoula, Montana 59812, USA

²Present address: USDA Forest Service, Pacific Northwest Research Station, 11175 Auke Lake Way, Juneau, Alaska 99801, USA

Full list of author information is available at the end of the article

Resumen

Antecedentes: Los tratamientos de reducción de combustibles han sido ampliamente usados a través del oeste de los EEUU en décadas recientes, tanto para protección de incendios como en restauración. Aunque las investigaciones han demostrado que la combinación de raleos y quemas prescriptas reducen efectivamente el potencial de tener fuegos de copas por algunos años después de estos tratamientos, pocas investigaciones han identificado la efectividad de los raleos y la posterior quema más allá de una década. Además, es poco claro como disturbios post- tratamientos tales como epidemias del escarabajo de la corteza afectan la efectividad de estos tratamientos.

Resultados: Evaluamos las diferencias en las características de combustibles superficiales y del dosel arbóreo y las propiedades potenciales de parámetros del comportamiento del fuego entre distintos tratamientos de reducción de combustible (ninguna acción o control, quema, raleo, y raleo+quema), implementado en bosques dominados por pino ponderosa (*Pinus ponderosa* Lawson & C. Lawson) y pino Oregón (*Pseudotsuga menziesii* [Mirb.] Franco), que fueron afectados por una epidemia del escarabajo del pino de montaña (*Dendroctonus ponderosae* Hopkins) después de efectuados estos tratamientos. Las unidades experimentales fueron medidas en 2002 (inmediatamente después de efectuados los tratamientos) y en 2016 (14 años post- tratamientos y al menos 4 años posteriores a la epidemia del escarabajo de montaña). Encontramos que los tratamientos de raleo afectados por el escarabajo de montaña (raleo, y raleo+quema) tenían menos combustible (*i.e.*, 34% y 83% menos de combustible fino y restos de troncos muertos en superficie), y menor potencial de fuegos de copas (*i.e.*, 47% menos probabilidad de coronamiento y 42% más de índice de coronamiento) que los correspondientes a los tratamientos no raleados (control y solo quema). No hubo otro efecto post- epidemia del escarabajo de montaña en los tratamientos de quema (solo quema y raleo+quema combinados) en los combustibles de superficie, aunque la quema redujo el potencial de coronamiento (*i.e.*, 37% más de índice de coronamiento) sobre las unidades no quemadas (control y solo raleadas) 14 años después del tratamiento. Adicionalmente, determinamos los impactos relativos de los tratamientos de combustible y la epidemia del escarabajo de montaña en las diferencias entre los combustibles y en el potencial de coronamiento. Encontramos que la mortalidad de árboles causada por el escarabajo de montaña aumentó las diferencias entre el tratamiento de control y los que involucraban tratamientos de raleos (raleo solo y raleo+quema), en lo que hace a la carga de combustible superficial y la probabilidad de coronamiento del fuego, pero disminuyó las diferencias entre esos tratamientos para la carga de combustible del dosel, su densidad aparente, y el índice de coronamiento.

Conclusiones: A pesar de los efectos diferenciales de las causales de muerte de árboles por el escarabajo de montaña en los tratamientos, nuestro estudio sugiere que los efectos de tratamientos de combustibles para mitigar el potencial de coronamiento de fuegos persiste aun después de la transformación de los rodales por la epidemia del escarabajo de montaña, especialmente cuando se combinan los tratamientos de raleos y quemas.

Background

A major goal of fuel reduction treatments is to mitigate potential wildfire behavior, especially to reduce the probability of crown fires in forests (Reinhardt *et al.* 2008). Many restoration efforts in fire-prone ecosystems also include fuel reduction strategies to reverse the effects of fire exclusion on forest fuels, structure, and composition (Brown *et al.* 2004, Fulé *et al.* 2012). However, treatment effects are ephemeral, as forest development in the years following treatment may compromise the effectiveness of treatments to mitigate crown fire behavior (Keyes and Varner 2006, Affleck *et al.* 2012, Tinkham *et al.* 2016). Wildland fire is not the only disturbance in many forests (Bebi *et al.* 2003, Bigler *et al.* 2005, Raffa *et al.* 2008, Kolb *et al.* 2016); fuel treatments may also interact with insect outbreaks and droughts, for example. Understanding how fuel treatments develop with time and in response to an exogenous disturbance such as a beetle

outbreak has important implications for fuel treatment effectiveness and longevity in light of management objectives.

Fuel treatments in forests are often designed to decrease crown fire behavior (*i.e.*, propensity for crown fire ignition or spread) by reducing canopy and surface fuels, and increase resistance to fire (*i.e.*, overstory survival) by favoring fire-tolerant species and removing competition. Fire exclusion and canopy ingrowth over the past century have elevated surface and canopy fuel loading in the western US (Parsons and DeBenedetti 1979, Covington and Moore 1994, Keeling *et al.* 2006). Increased surface and canopy fuel loading in conjunction with a warmer and drier climate have caused wildfires to increase in size and frequency, resulting in greater suppression costs to protect resources (Westerling *et al.* 2006, Flannigan *et al.* 2009, Miller *et al.* 2009). Crown fires threaten human safety and property, and, in forest types where

crown fire is uncharacteristic, it also threatens ecological resilience (Savage and Nystrom Mast 2005). Fuel reduction is a proactive silvicultural treatment that alters potential fire behavior by removing and modifying fuels to encourage low-severity (low overstory mortality) surface fire instead of high-severity (high overstory mortality) crown fire. Fuel treatments are typically designed to reduce surface fuel loading and canopy densities; increase heights to canopy base; and retain large, fire resistant trees (Agee and Skinner 2005, Hessburg *et al.* 2015). Although these goals can be attained with various silvicultural techniques, thinning and burning are the most typical means of fuel reduction. The relative effectiveness of thinning and burning to reduce crown fire behavior has been thoroughly studied immediately after treatment (Stephens and Moghaddas 2005, Harrington *et al.* 2007, Stephens *et al.* 2009, Fulé *et al.* 2012, McIver *et al.* 2012), generally highlighting that burning reduces surface fuels, thinning improves forest structure, and the combination of the two best reduces crown fire potential.

However, fuel treatments are only temporarily effective (Reinhardt *et al.* 2008, Martinson and Omi 2013). As treated areas age, regeneration, ingrowth, and residual trees grow into open space and increase surface and canopy fuel loading, causing concomitant increases in crown fire potential (Keyes and O'Hara 2002, Keyes and Varner 2006, Affleck *et al.* 2012). Although stimulated growth and regeneration are expected to follow treatment because of release from competition, it is still unclear how long treatments remain effective. Studies have identified that fuel treatments may be effective for a decade following treatment (Finney *et al.* 2005, Fernandes 2009, Jain *et al.* 2012, Stephens *et al.* 2012), but evidence beyond a decade is scant (though simulated by Tinkham *et al.* 2016). Understanding of treatment longevity is especially important when logistics and economics limit successive treatments.

Fire exclusion, past management actions, and warming climate have increased probability of crown fire in the West, but they have also been attributed to abetting recent insect outbreaks (Raffa *et al.* 2008, Bentz *et al.* 2010). Recent bark beetle outbreaks in the late 1990s to 2012 have profoundly affected several forest types, killing trees over millions of hectares in western US forests (Hicke *et al.* 2016). Wildfire in beetle-impacted forests is a primary concern for managers because beetle-killed trees alter canopy and surface fuel profiles (Page and Jenkins 2007, Hicke *et al.* 2012), as foliage on killed trees transitions from green to red to gray phases on the tree, then progressively falls to the forest floor with accompanying limbs and stems (British Columbia Ministry of

Forests 2004, Donato *et al.* 2013). The impact of beetle-caused tree mortality on potential fire behavior in unmanaged forest landscapes has been a controversial topic (Jenkins *et al.* 2008, 2014; Simard *et al.* 2011; Harvey *et al.* 2013; Hart *et al.* 2015; Kane *et al.* 2017).

Bark beetle outbreaks can also directly impact fuel treatments, further altering the fuel profile and fire hazard. The interactions between fuel treatments and beetle outbreaks remain largely uncharacterized. A few studies have identified that fuel treatments may moderate beetle-caused mortality (Fettig *et al.* 2010, Jenkins *et al.* 2014, Hood *et al.* 2016). Conversely, beetle-caused tree mortality may compromise fuel treatment effectiveness by altering surface fuel loading and vegetative competition, depending on time since outbreak initiation. In unmanaged units with beetle-killed trees that have lost their needles (*i.e.*, gray phase), active crown fire behavior and torching probability may be reduced because of altered canopy fuel attributes (Simard *et al.* 2011), similar to a fuel treatment. But beetle-caused mortality eventually adds fuel to the surface profile, and can thereby increase surface flame lengths, spotting, and residence times, exacerbating crown fire potential in surviving trees (Moran and Cochrane 2012, Jenkins *et al.* 2014). In treated units that densify and become more prone to crown fire over time, bark beetle-caused mortality may either maintain fuel treatment effectiveness by prolonging the duration of low canopy fuel loading and reducing canopy connectivity, or it may render treatments useless to their original objective by increasing surface fire behavior and probability of torching. Knowledge of fuels and potential crown fire behavior between treated and untreated units is valuable for safety assessment, inventory, and planning, as well as for determining resilience of actively managed units to subsequent disturbance.

The purpose of this study is to understand how silvicultural fuel reduction and a subsequent bark beetle outbreak influence fuel and the potential for crown fire. This study is distinct because it examines fuel treatments that were tested by a beetle outbreak with high levels of tree mortality rather than by low, endemic levels of bark beetle-caused tree mortality. We utilized the northern Rocky Mountains study site of the Fire and Fire Surrogate Study (McIver and Weatherspoon 2010) as a balanced experimental design to contrast fuel treatments (no-action control, burn-only, thin-only, thin+burn) in a (*Pinus ponderosa* Lawson & C. Lawson)–Douglas-fir (*Pseudotsuga menziesii* [Mirb.] Franco)-dominated forest. Treatments were fully implemented by 2002, at least four years before a widespread mountain pine beetle (MPB; *Dendroctonus ponderosae* Hopkins) outbreak that overlapped all experimental units. We analyzed surface

and canopy fuels data from 14 years after silvicultural treatment and at least 4 years after an MPB outbreak, with the specific objective to determine temporal effects of the combined silviculture and MPB outbreak on surface and canopy fuel characteristics. As a follow-up demonstration, we used an industry standard modeling approach (*i.e.*, using the Fire and Fuels Extension to the Forest Vegetation Simulator [FFE-FVS]) to predict crown fire potential given post-treatment conditions. Our final objective was to determine the individual effects of silvicultural treatment and the MPB outbreak on surface and canopy fuel characteristics and crown fire potential. This study uniquely showcases the impact that time and beetle outbreak have on restorative fuel treatments, demonstrating how beetle-caused mortality interacts with the development of fuel characteristics and crown fire potential in treated versus untreated units.

Methods

Study site

This study was conducted at the University of Montana's Lubrecht Experimental Forest (46°53'N, 113°26'W), an 11 300 ha forest in western Montana's Blackfoot River drainage of the Garnet Range, USA. Study sites ranged in elevation from 1230 to 1388 m ASL, and were composed of *Pseudotsuga menziesii*-*Vaccinium caespitosum* Michx. and *Pseudotsuga menziesii*-*Spiraea betulifolia* Pall. habitat types (Pfister et al. 1977). This forest was generally composed of second-growth ponderosa pine (*Pinus ponderosa* Lawson & C. Lawson var. *scopulorum* Engelm.), Douglas-fir (*Pseudotsuga menziesii* [Mirb.] Franco var. *glauca* [Beissn.] Franco), with western larch (*Larix occidentalis* Nutt.) regenerated from heavy cutting in the early twentieth century. Soils are fine or clayey-skeletal, mixed, Typic Eutroboralfs, as well as loamy-skeletal, mixed, frigid, Udic Ustochrepts (Nimlos 1986).

Climate in this study area is maritime-continental. Annual precipitation is approximately 460 mm (PRISM Climate Group, Oregon State University, <http://prism.oregonstate.edu>), nearly half of which falls as snow. Mean temperatures range from -6 °C in December and January to 17 °C in July and August. Average plant growing season is between 60 and 90 days. Historic fire frequency at Lubrecht prior to the twentieth century ranged from 2 to 14 years, with a mean composite fire return interval of 7 years (Grissino-Mayer et al. 2006).

Silvicultural activities and MPB "treatment"

A portion of the Lubrecht Experimental Forest was selected as a site for the Fire and Fire Surrogate Study, a multidisciplinary research project that aimed to quantify the short-term effects of restorative fuel treatments in frequent-fire forests across the US (McIver

and Weatherspoon 2010, Weatherspoon 2000). The study provided a framework to examine the effects of common fuel treatments on treatment longevity, fuel development, and potential fire behavior. Treatments were implemented in each of three blocks of approximately 36 ha (9 ha per treatment unit), using a randomized factorial design: two levels of thinning (thinned and unthinned) by two levels of prescribed burning (burned and unburned), for a total of four treatment levels (no-action control, burn-only, thin-only, thin+burn), with one treatment replicate per block. Prescription intensity was designed to maintain 80% overstory tree survival given a wildfire during 80th percentile weather conditions (Weatherspoon 2000). Units were cut in 2001 and burned in 2002, creating twelve 9 ha experimental units. The silvicultural cutting prescription was a combined low thinning and improvement cut to a residual basal area of 11.5 m² ha⁻¹, favoring retention of ponderosa pine and western larch over Douglas-fir (although Douglas-fir maintained a significant presence in residual overstories; Metlen and Fiedler 2006). Burning treatments were spring burns with wind speeds less than 13 km h⁻¹. Burns were generally low severity, with pockets of high severity in two of the thin+burn treatments. Fiedler et al. (2010) analyzed treatment effect on stand structure and short-term growth, and Stephens et al. (2009) summarized short-term woody fuel and potential fire behavior responses to treatment across western Fire and Fire Surrogate sites.

Approximately four growing seasons after treatment, a regional MPB outbreak in Montana began affecting all experimental blocks in the study (Gannon and Sontag 2010). Beetle-caused overstory mortality levels were high in the control and burn-only units from 2006 to 2012 (Hood et al. 2016), leading to comparable live ponderosa pine basal area across all treatments. After the outbreak, fuel loading, crown fire potential, productivity, and stand dynamics were no longer a pure effect of fuel treatments, but rather of the combination of fuel treatments and beetle-caused tree mortality. This beetle outbreak was an opportunity to assess a novel but increasingly common condition in the West: fuel treatment followed by a MPB outbreak. Therefore, the meaning of "treatment" in this study changes with measurement year. Before the MPB outbreak (prior to 2006), "treatment" refers to the silvicultural fuel treatment. After the MPB outbreak, "treatment" refers to the combination of fuel treatment and MPB-caused tree mortality.

Field methods

Live trees were measured twice on permanently monumented plots in the Fire and Fire Surrogate Study. Trees were initially measured the year after treatment (measured in 1999 for control, 2001 for thin-only, 2002 for burn-only and thin+burn) on 10 randomly selected plot locations from 36 systematically located grid points

within each treatment unit, for a total of 120 plots (Metlen and Fiedler 2006). Mature trees were measured on 0.04 ha circular subplots; for each mature tree with a diameter at breast height (dbh; 1.37 m) greater than 10.16 cm, species, dbh, and total height were recorded. Height to the base of live crown was recorded for all “leave” trees according to the silvicultural prescription (*i.e.*, all trees retained in the thin-only and thin+burn treatments, but only a similar subset of trees in the control and burn-only). Height to the base of live crown was estimated based upon the average branch height of the compacted lower limit of the crown (US Forest Service 2005). Trees smaller than 10.16 cm dbh but taller than 1.37 m were measured on five 100 m² subplots; trees between 0.10 m and 1.37 m tall were measured on twenty 1 m² subplots. Mature trees were re-measured in 2014, and trees smaller than 10.16 cm dbh were re-measured in 2016 using the protocol outlined above.

Dead surface fuel loading was first measured the year after treatment (same years as above: 1999 for control, 2001 for thin-only, 2002 for burn-only and thin+burn) using a mixture of planar intercept and destructive sampling. A modified Brown’s (1974) protocol was used to quantify 1-h (woody material <0.64 cm diameter), 10-h (0.64 cm ≤ diameter < 2.54 cm), 100-h (2.54 cm ≤ diameter < 7.62 cm), and 1000-h+ (diameter ≥ 7.62 cm) time-lag classes. On each of the 36 grid points, two 15.2 m transects were established; 1-h and 10-h fuels were tallied for 1.8 m of the length, 100-h fuels were tallied for 3.7 m, and 1000-h+ fuel diameters were recorded along the entire transect lengths. Duff and litter depths were each measured along transects at 4.6 m and 10.7 m from plot center. In the thin-only and thin+burn treatments, 1-h, 10-h, litter, and duff materials were not measured along transects but destructively sampled on two 1 m² quadrats. These materials were taken to the lab, oven dried, and weighed to determine loading by fuel component type. In 2016, we re-measured dead surface fuels using the original modified Brown’s transects for all 36 grid points in all of the treatment units.

For simplicity’s sake in this paper, the datasets will be referred to by the last year of measurement. Namely, “2002” for the collective immediate post-treatment dataset, and “2016” for the post-beetle-outbreak dataset. By the time of final measurement, units were in the post-epidemic, leaf-off, gray phase of the MPB rotation (Jenkins *et al.* 2008).

In addition to the 2002 and 2016 datasets, we supplemented our dataset with data measured and analyzed by Hood *et al.* (2016). Using the same measurement points described above, they measured MPB-caused tree mortality to the overstory (dbh > 10.16 cm) between 2006 and 2012. We augmented our dataset with their data on plot-scale MPB outbreak severity (overstory stems ha⁻¹).

Analytical and statistical methods

We calculated dead downed woody debris loading (Mg ha⁻¹) according to Brown (1974), and used site-specific depth-to-loading regressions to calculate litter and duff loading (Mg ha⁻¹) (M. Harrington, retired, USDA Forest Service, Missoula, Montana, USA, unpublished data). Dead downed woody fuel loading were grouped into two pools for analysis: fine woody debris (FWD) was composed of fuel less than 7.62 cm diameter (1-h, 10-h, and 100-h); coarse woody debris (CWD) was composed of sound fuel greater than or equal to 7.62 cm diameter (1000-h). Litter and duff layers (LD) were combined for analysis.

Surface fuels data and measured tree data were input into FFE-FVS (Dixon 2002, Rebain 2010) to calculate plot-scale canopy fuel characteristics and potential fire behavior for our two measurement years (2002, 2016). We estimated fire behavior using FFE-FVS methods, whereby measured dead fuel loadings are keyed to Albini’s (1976) 13 original fire behavior fuel models, and the FFE-FVS algorithm selects and weights predicted fire behavior from one to two most similar models (Rebain 2010). This default FFE-FVS method produces consistent results to predictions based entirely on user-inputted, customized fire behavior fuel models in FFE-FVS (Noonan-Wright *et al.* 2014). Potential fire behavior was based on FFE-FVS’s default “severe” fire weather scenario (4% 10-h fuel moisture, 21.1 °C ambient temperature, and 32.2 km h⁻¹ wind speed at 6.1 m) instead of percentile (*e.g.*, 80th or 95th) fire weather conditions to provide standardized analysis. Output gathered from FFE-FVS FUELOUT and POTFIRE reports included canopy fuel loading (CF; Mg ha⁻¹), canopy base height (CBH; m), canopy bulk density (CBD; kg m⁻³), potential fire behavior (fire type and surface flame length, m), and crown fire potential (probability of torching, PT, %; and crowning index, CI, km h⁻¹) calculations (see Rebain 2010 for further variable descriptions). We note that the CF loading from the FUELOUT report includes foliage plus branchwood < 7.62 cm diameter, but FFE-FVS only uses foliage plus half of all materials < 0.64 cm to calculate CBH and CBD (Rebain 2010).

We recognize that fire prediction models such as FFE-FVS have shortcomings and are simplified representations of actual fuel and weather conditions, especially in bark beetle-impacted forests. Cruz and Alexander (2010) discussed these limitations and found that models typically underpredict fire rate of spread in the forest canopy, and Fernandes (2009) identified that simulation for treatment comparison may produce conflicting results with actual fire behavior. There is no universally accepted alternative to modeling potential fire behavior over large spatial extents, although some researchers have developed empirical crown fire transition models with coarse data inputs (*e.g.*, Cruz *et al.* 2005) or novel

approaches to address FFE-FVS's shortcomings (e.g., Nelson *et al.* 2017). Computational fluid dynamics models have been used to examine potential fire behavior in beetle-impacted forests (Sieg *et al.* 2017), but these models are not feasible to use on the large, 9 ha treatment units. Despite these shortcomings, FFE-FVS is the standard fuel and fire assessment tool used by vegetation managers, planners, and researchers because it extends fuels data to more meaningful management objectives. Therefore, we used FFE-FVS as a tool to demonstrate treatment-specific outcomes for mitigating fire hazard as it is commonly assessed and given model assumptions. This is a useful means for contrasting treatment alternatives (Johnson *et al.* 2011); however, we caution users that these predictions may not reflect actual fire behavior, a critical consideration for actual management applications.

We used nested ANOVA to investigate treatment influence on eight fuel and crown fire potential response variables in 2002 and 2016. In this study, plot is nested within experimental unit, which is nested within block. We performed this analysis using the `anova.lme` function in R's `nlme` package (Pinheiro *et al.* 2016, R Core Team 2016). ANOVA models had the form:

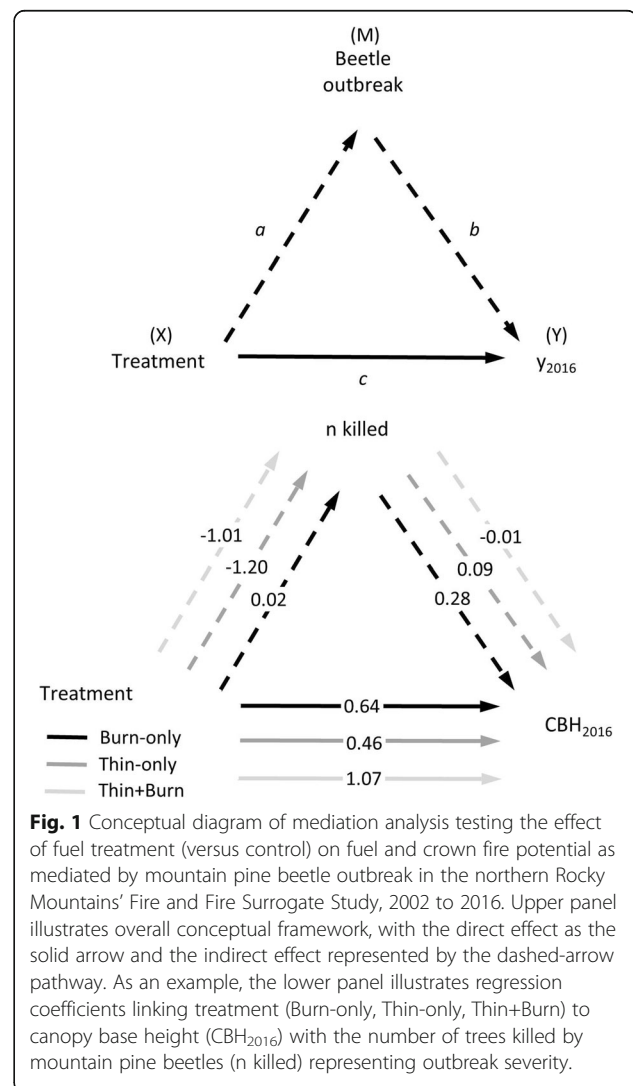
$$\hat{y}_{ijkl} = \mu + \alpha_i + \beta_j \times \gamma_k + \varepsilon_{ijk} + \delta_{ijkl} \quad (1)$$

where $\hat{y}$ is the plot-scale response variable (2002 and 2016 FWD loading, CWD loading, LD loading, CF loading, CBH, CBD, PT, and CI), μ is the grand mean, α_i is the block effect (levels 1 to 3), β_j is the prescribed burn effect (levels not burned and burned), γ_k is the thinning effect (levels not thinned and thinned), ε_{ijk} is the experimental unit error term, and δ_{ijkl} is the residual error term associated with plots. Although the block effect would ideally be treated as a random effect, we considered it a fixed effect in this model because there were only three factor levels; therefore, only experimental unit was treated as a random effect.

Next, we used linear mixed effects regression to determine the effect that treatment has on the development of fuel loading and crown fire potential over time (i.e., change in value from 2002 to 2016) using the `lme` function in `nlme`. Regression models had the same structure as the nested ANOVA model, except that $\hat{y}$ is the change in plot-scale response variable (FWD, CWD, LD, CF, CBH, CBD, PT, and CI) from 2002 to 2016.

Finally, we conducted mediation analyses to parse out the effects of silvicultural manipulation and the MPB outbreak on fuel and crown fire potential metrics. Mediation analysis allows insight into direct treatment effects and the indirect effects mediated by MPB, as, by 2016, the fuel complex in the treatments was a function of both the fuel treatment and the MPB outbreak. The goal

was to characterize the direct effect of the silvicultural treatments (X) on eight different 2016 fuel and crown fire potential metrics (Y), the indirect effect of the treatments on the metrics as mediated by the outbreak (M), and the total effect of the treatments on the metrics given mediation by the outbreak (Fig. 1; Baron and Kenny 1986, MacKinnon *et al.* 2007). Coefficients are derived by fitting two statistical models: $Y = f(X, M)$, and $M = f(X)$. The direct effect is quantified as the regression coefficient of the relationship between X and Y (leg c of Fig. 1), the indirect effect is quantified as the product of the relationships between X and M (leg a of Fig. 1) and M and Y (leg b of Fig. 1), while the total effect is the sum of direct and indirect effects. We determined the relationships a , b , and c using linear mixed effects regression with the same nesting structure characterized in our ANOVA models. Since we wanted to determine the effect that treatment had on mediation, we contrasted each of the active treatment effects with the



control (*i.e.*, burn-only vs. control, thin-only vs. control, thin+burn vs. control). All variables were standardized for interpretation of effect size across fuel and crown fire potential metrics, with effect sizes near zero meaning no difference from the control treatment. The bottom panel of Fig. 1 illustrates standardized coefficients for CBH as an example. We utilized a non-parametric bootstrap re-sampling routine ($N = 1000$ replications) to determine if direct, indirect, and total effects were significantly different from zero.

In all analyses, treatment effects were considered to have strong evidence of significance at the 95% confidence level, and marginal evidence of significance at the 90% level. We inspected residuals from nested ANOVA and linear mixed effects regression models of response state and change for constant variance across treatments using Levene's test of homoscedasticity. When residuals were heteroscedastic, we applied treatment level variance functions using R's `varIdent` function. Furthermore, we applied square root transformations on responses that showed increasing residual variance with predicted values.

When expedient for summarizing broad patterns and concise interpretation, we grouped treatments according to the crossed factorial design nomenclature. Thinned or thinning refers to thin-only and thin+burn treatments, while unthinned refers to control and burn-only. Burned or burning refers to burn-only and thin+burn treatments, while unburned refers to control and thin-only.

Results

Thinning was the dominant influence on overstory stand structure and development across measurement years (Table 1), influencing the fuel and fire behavior responses in this study. Thinned units had 67% lower stem densities than unthinned units in 2002 (79% and 60% lower by basal area and stand density index, respectively), but all density metrics were more similar across treatments by 2016. Although differences between

thinned and unthinned units in stem density, basal area, and stand density index abated over time by 33%, 63%, and 54%, respectively, the contrast between thinned and unthinned quadratic mean diameters increased by 161% over the measurement period as large trees in the unthinned units were killed by MPB.

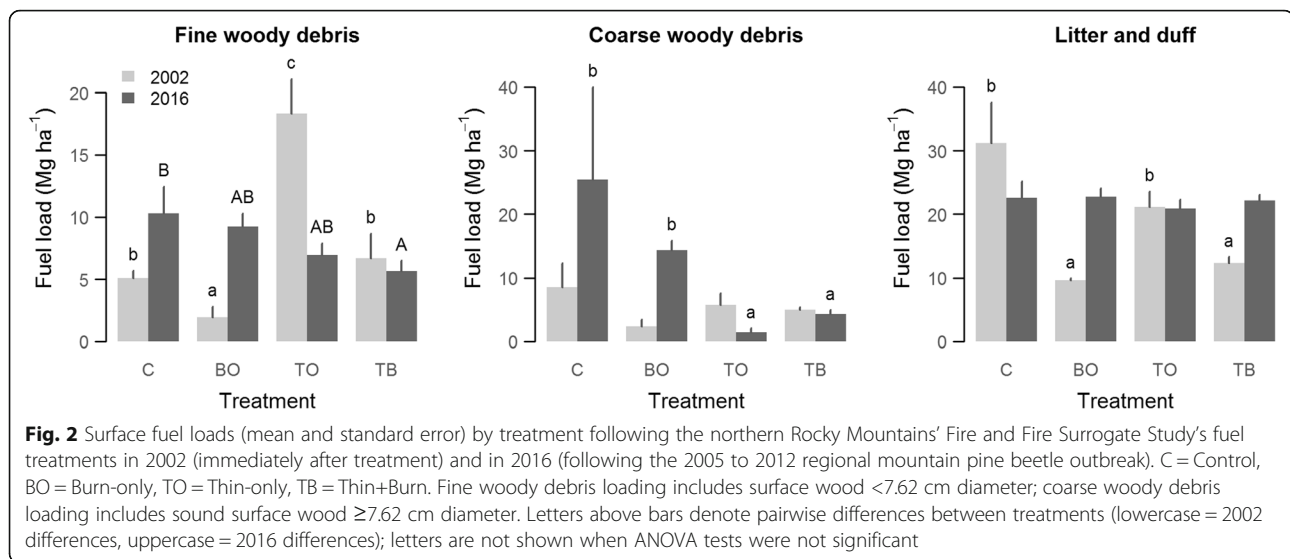
Surface fuel loading

Treatment effect on FWD loading in 2002 (the year following silvicultural treatment) followed an expected pattern (Fig. 2). Burning, thinning, and their interaction all had significant effects on FWD (Table 2; $P \leq 0.024$). Burning reduced FWD loading by 63% (compared to unburned) and thinning increased FWD loading by 250% (compared to unthinned). In 2016 (at least 4 years following MPB-caused mortality), only thinning had a significant effect, with 34% less FWD in thinned than unthinned treatments (Table 2; $P = 0.028$). Whereas in 2002, thin+burn and control loading were no different, in 2016 these two were the only individual treatments that were statistically distinct (Fig. 2). Unthinned units significantly accumulated fuel between 2002 and 2016 (Table 3; $P \leq 0.007$), but FWD in the thin-only treatment decreased ($P = 0.010$) and did not change in the thin+burn treatment ($P = 0.490$).

CWD loading was similar across treatments in 2002 (Fig. 2; Table 2), but by 2016, CWD loading was lower in thinned than unthinned treatments (Table 2; $P = 0.002$). Variability of CWD loading among control units (*i.e.*, standard deviation) was 15 times greater than treated units because of one unit with particularly high CWD loading. Overall, thinned treatment CWD loading was 83% less than the unthinned treatments. Similar to trends observed in FWD dynamics, unthinned treatments accumulated CWD from 2002 to 2016 (Table 3; $P \leq 0.057$), whereas CWD in the thin-only treatment reduced over time (Table 3; $P = 0.015$) and the thin+burn treatment did not change ($P = 0.601$).

Table 1 Mean (and standard error) stand structure metrics by treatment following the Fire and Fire Surrogate Study's fuel treatments in 2002 (immediately after treatment) and in 2016 (following the 2005 to 2012 regional mountain pine beetle outbreak). Stand density index was calculated according to Reineke (1933), equivalent to 25.4 cm trees per hectare

Year	Treatment	Density (stems ha ⁻¹)	Basal area (m ² ha ⁻¹)	Stand density index (metric)	Quadratic mean diameter (cm)
2002	Control	322 (60)	23.1 (5.7)	420 (99)	31.1 (1.5)
2002	Burn-only	304 (53)	20.8 (2.7)	384 (51)	31.2 (1.0)
2002	Thin-only	111 (11)	9.7 (0.4)	170 (8)	33.4 (0.5)
2002	Thin+Burn	94 (12)	8.8 (1.2)	151 (22)	33.4 (0.9)
2016	Control	323 (60)	17.8 (5.1)	339 (91)	26.7 (1.9)
2016	Burn-only	291 (43)	17.5 (2.9)	328 (49)	28.0 (2.4)
2016	Thin-only	188 (27)	14.9 (0.4)	265 (11)	33.4 (2.1)
2016	Thin+Burn	99 (13)	10.9 (0.8)	181 (15)	33.0 (1.5)



LD loading varied by treatment in 2002 (Fig. 2; Table 2). Loads were 59% lower in burned treatments than unburned treatments ($P = 0.007$). Although ANOVA results indicated a significant burning and thinning interaction ($P = 0.038$), pairwise comparisons show LD loading assembles into two main treatment groups: burned and unburned. In 2016, LD loading did not vary by treatment (Table 2). Burned treatments significantly accumulated LD loading between 2002 and 2016 (Table 3; $P < 0.006$), but LD in unburned treatments either decreased (control; $P = 0.083$) or did not change ($P = 0.9288$).

Canopy fuel characteristics

CF loading differed by thinning in 2002 (Fig. 3; Table 2). The immediate effect of thinning was a 58% reduction in CF versus the unthinned treatments ($P = 0.001$). In 2016, there was slight evidence of both thinning and burning effects (Table 2; P of 0.096 and 0.060, respectively). These effects were relatively minor on their own, but when combined, caused the thin+burn treatment to have 43% less CF than the control. Thinned treatment CF loading increased between 2002 and 2016 ($P \leq 0.018$), but unthinned treatment loading decreased ($P \leq 0.049$).

CBH in 2002 varied due to thinning (Fig. 3; Table 2), whereby mean CBHs were 130% higher in thinned treatments than unthinned ($P < 0.001$). By 2016, CBH varied by both burning and thinning (Table 2). Burned treatments were associated with 105% greater CBHs than unburned ($P = 0.008$), and thinned treatments had 79% greater CBHs than unthinned treatments ($P = 0.008$). There was slight evidence that interaction amplified these effects ($P = 0.094$) such that the thin+burn treatment was 3.2 times greater than the control. CBH dropped significantly in the thin-only treatment from 2002 to 2016 (Table 3; $P = 0.003$) as ladder fuel ingrowth

densified the canopy from below, but CBH did not change in the remaining treatments ($P \geq 0.350$).

Immediately after treatment (*i.e.*, in 2002), CBD in thinned treatments was 55% less than in unthinned treatments (Fig. 3; Table 2; $P < 0.001$). In 2016, CBDs were more similar among treatments than in 2002, but still varied significantly by treatment (Table 2). Burned treatments had 33% lower CBDs than unburned treatments ($P = 0.0314$), and thinned treatments had 46% lower CBDs than unthinned treatments ($P = 0.004$). Although change in CBD between 2002 and 2016 appears to vary by thinning (reduction in unthinned due to MPB-caused tree mortality and accumulation in thinned due to ingrowth), reduction was only significant for the burn-only treatment (Table 3; $P = 0.040$) and accumulation for the thin-only treatment ($P = 0.008$).

Surface and crown fire potential

The FFE-FVS software keyed at least one fire behavior fuel model to each plot, but fuel model 8 ("closed timber litter") was the most commonly assigned model across 2002 treatments (Table 4). Fuel model 8 was also the most assigned model in thinned treatments in 2016, but the unthinned treatments were better characterized by fuel model 10 ("timber [litter and understory]"), with occasional assignments of fuel model 12 ("medium logging slash"). Predicted surface fire flame length was greatest in the thin-only treatment and lowest in the burn-only treatment in 2002, but the thin-only treatment had the lowest predicted flame lengths in 2016. In 2002, crown fire (passive type) was only predicted for the burned treatments (13% of plots). Passive crown fire was predicted for all treatment types in 2016. However, the control had the greatest propensity by far for crown fire, whether active, passive, or conditional.

Table 2 Nested ANOVA of fuel and crown fire potential by treatment following the Fire and Fire Surrogate Study's fuel treatments in 2002 (immediately after treatment) and in 2016 (following the 2005 to 2012 regional mountain pine beetle outbreak). NumDF and DenDF are the numerator and denominator degrees of freedom used in ANOVA, respectively. ^aFine woody debris (surface wood < 7.62 cm diameter); ^bCoarse woody debris (sound surface wood ≥ 7.62 cm diameter); ^cLitter and duff (litter and duff layers); ^dCanopy fuels (foliage and materials < 7.62 cm diameter); ^eCanopy base height (lowest height where canopy bulk density exceeds 0.011 kg m⁻³); ^fCanopy bulk density (maximum canopy fuel mass per volume given 4.5 m running mean); ^gProbability of torching (probability of surface fire ascending into crowns given Monte Carlo simulation); ^hCrowning index (6.1 m wind speed required to cause active crown fire)

Response	Source	NumDF	DenDF	2002		2016	
				<i>F</i> -statistic	<i>P</i> -value	<i>F</i> -statistic	<i>P</i> -value
FWD ^a (Mg ha ⁻¹)	Intercept	1	108	108.85	<0.0001	714.8	<0.0001
	Block	2	6	5.93	0.0380	1.33	0.3320
	Burning	1	6	24.83	0.0025	1.28	0.3018
	Thinning	1	6	32.56	0.0013	8.38	0.0275
	Thinning × Burning	1	6	8.94	0.0243	0.11	0.7527
CWD ^b (Mg ha ⁻¹)	Intercept	1	108	71.79	<0.0001	90.25	<0.0001
	Block	2	6	0.80	0.4914	0.54	0.6077
	Burning	1	6	2.95	0.1369	0.18	0.6890
	Thinning	1	6	1.44	0.2758	26.93	0.0020
	Thinning × Burning	1	6	1.17	0.3203	2.31	0.1792
LD ^c (Mg ha ⁻¹)	Intercept	1	108	306.12	<0.0001	1916.24	<0.0001
	Block	2	6	0.77	0.5028	0.74	0.5160
	Burning	1	6	39.95	0.0007	0.12	0.7420
	Thinning	1	6	3.93	0.0949	0.20	0.6674
	Thinning × Burning	1	6	7.00	0.0382	0.22	0.6538
CF ^d (Mg ha ⁻¹)	Intercept	1	108	276.03	<0.0001	284.01	<0.0001
	Block	2	6	1.07	0.3999	2.04	0.2109
	Burning	1	6	1.69	0.2409	5.33	0.0604
	Thinning	1	6	46.79	0.0005	3.89	0.0959
	Thinning × Burning	1	6	0.01	0.9356	1.16	0.3225
CBH ^e (m)	Intercept	1	105	345.6	<0.0001	258.37	<0.0001
	Block	2	6	0.99	0.4251	3.73	0.0884
	Burning	1	6	3.03	0.1323	15.03	0.0082
	Thinning	1	6	60.38	0.0002	15.67	0.0075
	Thinning × Burning	1	6	1.83	0.2247	3.96	0.0938
CBD ^f (kg m ⁻³)	Intercept	1	105	315.84	<0.0001	1674.82	<0.0001
	Block	2	6	0.44	0.6636	3.23	0.1118
	Burning	1	6	0.82	0.4012	7.81	0.0314
	Thinning	1	6	52.78	0.0003	20.16	0.0041
	Thinning × Burning	1	6	0.02	0.9003	0.00	0.9467
PT ^g (%)	Intercept	1	105	38.18	<0.0001	151.13	<0.0001
	Block	2	6	0.13	0.8768	4.32	0.0688
	Burning	1	6	22.85	0.0031	0.85	0.3920
	Thinning	1	6	15.81	0.0073	18.73	0.0049
	Thinning × Burning	1	6	0.57	0.4775	0.40	0.5510

Table 2 Nested ANOVA of fuel and crown fire potential by treatment following the Fire and Fire Surrogate Study's fuel treatments in 2002 (immediately after treatment) and in 2016 (following the 2005 to 2012 regional mountain pine beetle outbreak). NumDF and DenDF are the numerator and denominator degrees of freedom used in ANOVA, respectively. ^aFine woody debris (surface wood < 7.62 cm diameter); ^bCoarse woody debris (sound surface wood ≥ 7.62 cm diameter); ^cLitter and duff (litter and duff layers); ^dCanopy fuels (foliage and materials < 7.62 cm diameter); ^eCanopy base height (lowest height where canopy bulk density exceeds 0.011 kg m⁻³); ^fCanopy bulk density (maximum canopy fuel mass per volume given 4.5 m running mean); ^gProbability of torching (probability of surface fire ascending into crowns given Monte Carlo simulation); ^hCrowning index (6.1 m wind speed required to cause active crown fire) (Continued)

Response	Source	NumDF	DenDF	2002		2016	
				F-statistic	P-value	F-statistic	P-value
CI ^h (km h ⁻¹)	Intercept	1	105	704.41	<0.0001	306.23	<0.0001
	Block	2	6	2.45	0.167	1.49	0.299
	Burning	1	6	1.80	0.2288	7.14	0.0369
	Thinning	1	6	54.16	0.0003	8.79	0.0251
	Thinning × Burning	1	6	0.01	0.9294	1.44	0.2754

PT in 2002 depended on both burning and thinning treatments (Fig. 4; Table 2). Burning reduced PT by 64% ($P = 0.003$) and thinning reduced it by 34% ($P = 0.007$). Probability of torching in 2016 was only dependent on thinning (Table 3). In this case, unthinned treatment PT was 111% greater than thinned treatments probabilities ($P = 0.005$). Between 2002 and 2016, PT in unthinned treatments significantly increased with inputs to the surface fuel loading (Table 3; $P \leq 0.036$), but PT did not significantly change in thinned treatments ($P \geq 0.124$).

CI differed by thinning in 2002 (Fig. 4; Table 2), which was expected because thinning reduced CBDs. More specifically, thinning resulted in 95% greater CIs than the unthinned treatments. By 2016, CI differed by both burning and thinning (Table 2). Burned units had 37% greater CI than unburned units ($P = 0.037$) and thinned units had 42% greater CI than unthinned units ($P = 0.025$). Although the ANOVA interaction term was not significant, pairwise differences revealed that the thin+burn treatment had 48% to 89% greater CI than the remaining treatments. CIs remained relatively constant between 2002 and 2016 except for in the thin-only treatment, where it dropped significantly as ingrowth densified the canopy (Table 3; $P = 0.019$).

Fuel and crown fire potential mediation

Differences in fuel loading between control and thinning treatments were mediated by the MPB outbreak, but the outbreak did not affect differences in loading between the control and burn-only treatments (Fig. 5). The significantly non-zero indirect effects in the thin-only and thin+burn for FWD, CWD, and CF responses demonstrate that the MPB outbreak mediated differences between control and thinned treatments for three of our four fuel loading metrics. Substantial (standardized effect size $P > 0.05$) indirect effects were consistent with total

effect direction for all responses except for CF, meaning that the MPB outbreak generally increased contrasts between control and treatments, but decreased CF differences between control and thinned treatments.

Consistent with fuel loadings, differences in CBH, CBD, and crown fire potential metrics between control and thinning treatments depended on the MPB outbreak, but differences between control and burn-only did not (Fig. 5). Thin-only indirect effects were significantly non-zero for CBD, PT, and CI, while thin+burn indirect effects were only significant for CBD and PT. Thus, the MPB outbreak did not affect CBH differences between control and treatments, nor did it affect any response differences between control and burn-only. Indirect effects were consistent with total effect directions only for PT; they were inconsistent with total effects for CBH, CBD, and CI. The magnitude and direction of indirect effects on PT illustrate that most of the difference between the control and thinned treatments was due to the MPB outbreak. Conversely, the MPB outbreak obscured differences in CBH, CBD, and CI between control and thinned treatments.

Discussion

This study characterized fuel development and crown fire potential dynamics 14 years after initial treatment and at least 4 years following an MPB outbreak. In general, we observed that fuel loading was elevated after the outbreak and ingrowth, and potential for crown fire was greatest in the untreated control, intermediate in burn-only and thin-only, and lowest in thin+burn (Table 5).

Despite the subsequent biotic disturbance, the primary management objective in these treated units is still to resist crown fire. Interaction between disturbances such as a beetle outbreak and fire has been a growing concern (Bebi et al. 2003, Schoennagel et al. 2012, Jenkins et al. 2014,

Table 3 Fuel and crown fire potential change by treatment between 2002 (immediately following treatment) and 2016 (following the 2005 to 2012 regional mountain pine beetle outbreak) at the Fire and Fire Surrogate Study. Estimates were derived and tested against zero using linear mixed effects models. ^aFine woody debris (surface wood < 7.62 cm diameter); ^bCoarse woody debris (sound surface wood ≥ 7.62 cm diameter); ^cLitter and duff (litter and duff layers); ^dCanopy fuels (foliage and materials < 7.62 cm diameter); ^eCanopy base height (lowest height where canopy bulk density exceeds 0.011 kg m⁻³); ^fCanopy bulk density (maximum canopy fuel mass per volume given 4.5 m running mean); ^gProbability of torching (probability of surface fire ascending into crowns given Monte Carlo simulation); ^hCrowning index (6.1 m windspeed required to cause active crown fire)

Response	Treatment	Estimate	Standard error	P-value
FWD ^a (Mg ha ⁻¹)	Control	5.17	1.30	0.0073
	Burn-only	7.32	1.35	0.0016
	Thin-only	-11.39	3.04	0.0095
	Thin+Burn	-1.02	1.39	0.4904
CWD ^b (Mg ha ⁻¹)	Control	16.96	7.20	0.0565
	Burn-only	12.02	2.99	0.0070
	Thin-only	-4.34	1.29	0.0152
	Thin+Burn	-0.71	1.29	0.6007
LD ^c (Mg ha ⁻¹)	Control	-8.59	4.14	0.0832
	Burn-only	13.14	2.28	0.0012
	Thin-only	-0.25	2.66	0.9288
	Thin+Burn	9.82	2.32	0.0055
CF ^d (Mg ha ⁻¹)	Control	-7.61	3.09	0.0488
	Burn-only	-7.06	2.02	0.0128
	Thin-only	9.47	0.76	<0.0001
	Thin+Burn	3.58	1.11	0.0177
CBH ^e (m)	Control	-0.34	0.53	0.5479
	Burn-only	0.53	0.52	0.3498
	Thin-only	-2.54	0.52	0.0028
	Thin+Burn	-0.31	0.55	0.5941
CBD ^f (kg m ⁻³)	Control	-0.006	0.007	0.4431
	Burn-only	-0.017	0.007	0.0404
	Thin-only	0.015	0.004	0.0080
	Thin+Burn	0.005	0.004	0.2406
PT ^g (%)	Control	20.46	7.61	0.0361
	Burn-only	32.97	7.50	0.0046
	Thin-only	4.22	7.50	0.5938
	Thin+Burn	14.02	7.83	0.1237
CI ^h (km h ⁻¹)	Control	5.84	8.33	0.5096
	Burn-only	11.41	8.19	0.2130
	Thin-only	-25.95	8.19	0.0193
	Thin+Burn	0.77	8.63	0.9318

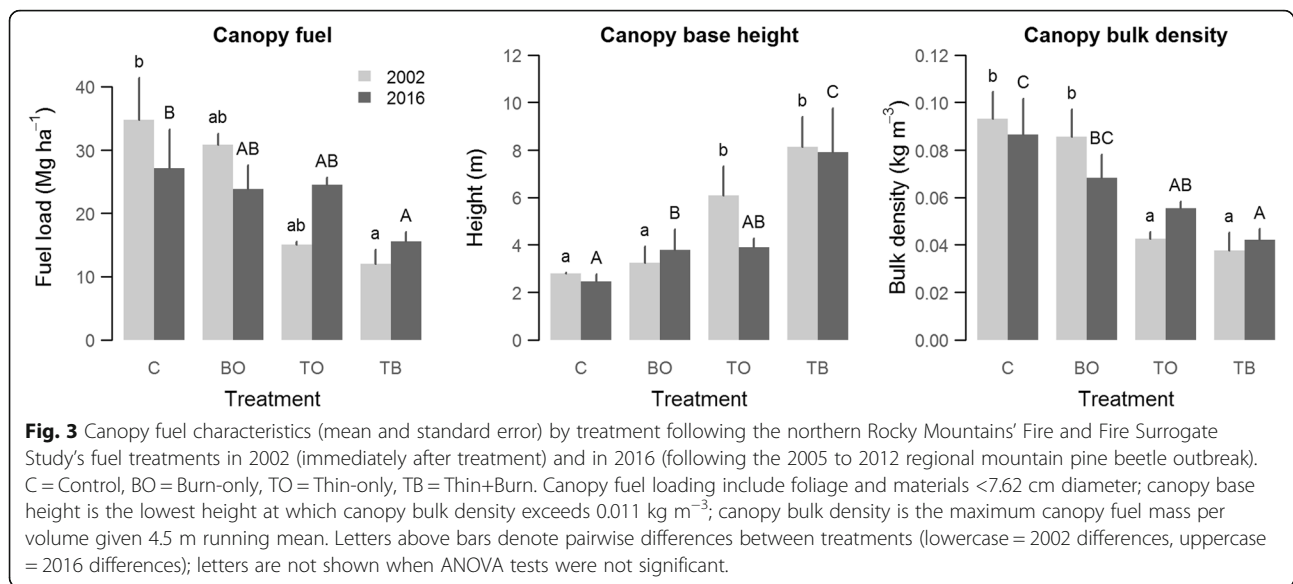
Kane *et al.* 2017), but other studies have not specifically considered this interaction within treated areas. To more satisfactorily address the drivers and outcomes of combined treatment and MPB-caused tree mortality effects, we discuss the mediation analysis prior to the assessment of fuel and crown fire potential dynamics.

Are differences driven by silvicultural treatment or MPB?

To effectively interpret treatment outcomes we must begin with the relationship between the two “treatment” components in 2016: silvicultural fuel reduction and MPB-caused tree mortality. Studies have shown endemic (non-outbreak) beetle populations can cause additional mortality in response to burning treatments, killing injured or less vigorous trees that may have survived burning alone (e.g., Larsson *et al.* 1983, Negrón and Popp 2004, Fettig *et al.* 2010). On our sites, Six and Skov (2009) identified that, by 2008, three bark beetle species (Douglas-fir beetle [*Dendroctonus pseudotsugae* Hopkins], pine engraver [*Ips pini* Say], and western pine beetle [*Dendroctonus brevicornis* LeConte]) increased in abundance because of burning treatments. MPB population size did not respond to treatment, but successful MPB attacks were more prevalent in unthinned treatments. By 2012, the regional MPB outbreak had caused high overstory mortality in the control (50%) and burn-only (39%) treatment units, leading to similar live ponderosa pine basal areas across all treatments (Hood *et al.* 2016).

We applied these MPB-caused tree mortality data to our mediation analysis, confirming that fuel treatment and MPB-caused mortality were inextricably linked: the number of overstory trees killed had a strong negative association with thinning and a slight positive association with burning. In addition to characterizing the combined effects of these “treatments,” our analysis ascertains the relative effects of silviculture and MPB on forest fuels and crown fire potential, including treatment-outbreak agreement or antagonism (Fig. 5; Table 5).

A number of studies have shown that, in unmanaged units, MPB-caused mortality alters forest fuel profiles (summarized in Jenkins *et al.* 2012, but see Simard *et al.* 2011). Our mediation analysis illustrates that CF loading after the outbreak was significantly lower in thin+burn than control despite high MPB-caused tree mortality in control units, and FWD loading was lower in thin+burn units because of the high MPB-caused tree mortality. Additionally, MPB-caused tree mortality inflated differences between thinned and control units in FWD and CWD pools, but reduced existing differences in the CF pool. These linked fuel loadings demonstrate that MPB caused fuel transfer from overstory to surface pools, and that although MPB-caused tree mortality partially masked or diminished differences between thinned and unthinned



canopies, unthinned canopy fuels translocated to the ground inflated surface fuel loading beyond thinned treatments. The nature of these differences is also manifested in the divergence of assigned fire behavior fuel models by treatments. FFE-FVS assigned slash group fire behavior fuel models ("medium logging slash" and "heavy logging slash") to characterize unthinned surface fuel profiles, which is expected to make potential fire behavior more volatile, increasing soil heating and belowground severity.

Although MPB-caused tree mortality inflated surface fuel differences between thinned and control units, our analysis of fire potential indicated MPB only inflated differences in PT between those treatments. Studies have shown that an MPB outbreak can exacerbate fire behavior, depending on

time since disturbance and metrics analyzed (synthesized in Jenkins *et al.* 2014, but see Harvey *et al.* 2013). But beyond beetle-caused inputs to surface fuels, beetles actually thin out forest canopies and eventually moderate potential crown fire spread, akin to the silviculturist's fuel treatments. Our analysis shows that MPB-caused tree mortality—which was greater in unthinned units—offsets the initial positive effect of thinning on canopy fuel (CBD) and therefore crown fire potential (CI). However, since this offset effect is minor (indirect effect magnitude was smaller than direct effect magnitude), it demonstrates that "natural thinning" by MPB neither reduces crown fire potential as well as active management nor hinders the effective longevity of silvicultural thinning. PT, on the other hand, incorporates

Table 4 Dominant fuel models and potential fire behavior by treatment following the Fire and Fire Surrogate Study's fuel treatments in 2002 (immediately after treatment) and in 2016 (following the 2005 to 2012 regional mountain pine beetle outbreak). Fuel models and fire behavior were determined using the Fire and Fuels Extension of the Forest Vegetation Simulator (FFE-FVS) Predicted fire behavior is based on standard severe fire weather conditions (FFE-FVS "severe" category: 4% 10-h fuel moisture, 21.1 °C, and 32.2 km h⁻¹ 6.1 m wind speed. Fuel model numbers refer to those developed by Albini (1976). Predicted fire type refers to categories outlined by Scott and Reinhardt (2001)

Year	Treatment	Primary fuel model				Surface flame length (m)	Predicted fire type			
		8	10	12	13		Surface	Conditional	Passive	Active
		Percent occurrence					Percent occurrence			
2002	Control	67 (3)	27 (7)	7 (3)	0 (0)	0.83 (0.01)	63 (13)	23 (12)	13 (3)	0 (0)
2002	Burn-only	93 (3)	7 (3)	0 (0)	0 (0)	0.43 (0.06)	80 (10)	20 (10)	0 (0)	0 (0)
2002	Thin-only	53 (3)	30 (0)	17 (3)	0 (0)	1.54 (0.03)	87 (3)	0 (0)	13 (3)	0 (0)
2002	Thin+Burn	93 (3)	3 (3)	3 (3)	0 (0)	0.71 (0.13)	100 (0)	0 (0)	0 (0)	0 (0)
2016	Control	33 (20)	40 (15)	23 (19)	3 (3)	1.56 (0.32)	40 (15)	17 (12)	40 (15)	3 (3)
2016	Burn-only	37 (9)	50 (10)	13 (3)	0 (0)	1.50 (0.13)	70 (12)	7 (7)	23 (7)	0 (0)
2016	Thin-only	70 (6)	30 (6)	0 (0)	0 (0)	1.11 (0.03)	83 (3)	0 (0)	17 (3)	0 (0)
2016	Thin+Burn	60 (0)	40 (0)	0 (0)	0 (0)	1.47 (0.11)	90 (6)	0 (0)	10 (6)	0 (0)

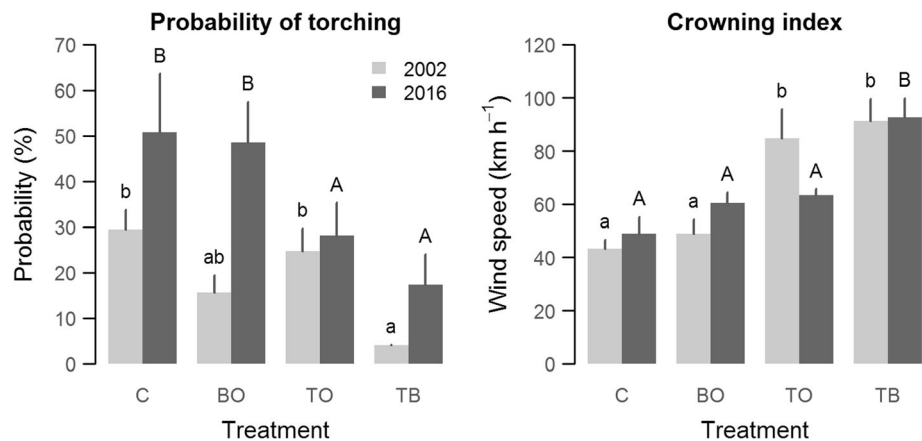


Fig. 4 Crown fire potential (mean and standard error) by treatment following the northern Rocky Mountains' Fire and Fire Surrogate Study's fuel treatments (completed in 2001) and regional mountain pine beetle outbreak (2005 to 2012). C = Control, BO = Burn-only, TO = Thin-only, TB = Thin+Burn. Probability of torching is the probability of surface fire ascending into crowns given Monte Carlo simulation; crowning index is the 6.1 m wind speed required to cause active crown fire. Letters above bars denote pairwise differences between treatments (lowercase = 2002 differences, uppercase = 2016 differences); letters are not shown when ANOVA tests were not significant.

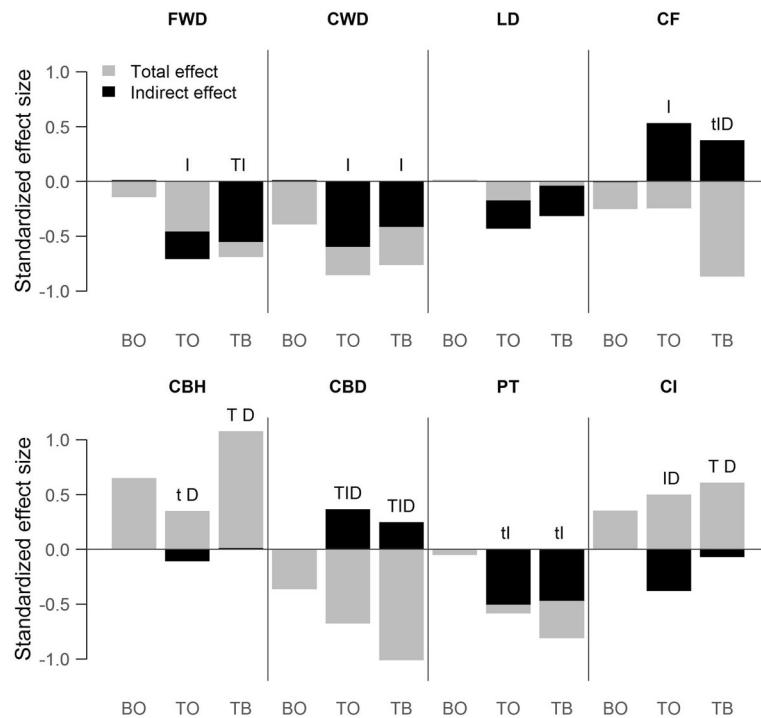


Fig. 5 Mediation analysis treatment effect sizes (vs. Control) on 2016 fuel (top panel) and crown fire potential (bottom panel) after the northern Rocky Mountains' Fire and Fire Surrogate Study. BO = Burn-only, TO = Thin-only, TB = Thin+Burn. Total effect represents observed or calculated treatment effect, indirect effect represents influence of treatment mediated by mountain pine beetle outbreak on total effect, and direct effect (total minus indirect) represents stand-alone treatment effect. Effect significance at 95% confidence level is shown by capital letter above bars (T = total, I = indirect, D = direct); lowercase letters signify significance at 90% confidence. Response variables include: FWD = fine woody debris, CWD = coarse woody debris, LD = litter and duff, CF = canopy fuels, CBH = canopy base height, CBD = canopy bulk density, PT = probability of torching, CI = crowning index.

Table 5 Summary of fuel load and crown fire potential differences by treatment at the Fire and Fire Surrogate Study. Fuel and crown fire potential attributes: FWD = fine woody debris load, CWD = coarse woody debris loading, LD = litter and duff loading, CF = canopy fuel loading, CBH = canopy base height, CBD = canopy bulk density, PT = probability of torching, CI = crowing index

Treatment	2002 to 2016 change ^a							
	FWD	CWD	LD	CF	CBH	CBD	PT	CI
Control	↑	↑	↓	↓			↑	
Burn-only	↑	↑	↑	↓		↓	↑	
Thin-only	↓		↓	↑	↓	↑		↓
Thin+Burn			↑	↑				
	Beetle outbreak impact (vs. Control) ^b							
	FWD	CWD	LD	CF	CBH	CBD	PT	CI
Control								
Burn-only								
Thin-only	↔	↔		→←		→←	↔	→←
Thin+Burn	↔	↔		→←		→←	↔	
	2016 Tukey-Kramer rank ^c							
	FWD	CWD	LD	CF	CBH	CBD	PT	CI
Control	B	B		B	A	C	B	A
Burn-only	AB	B		AB	B	BC	B	A
Thin-only	AB	A		AB	AB	AB	AB	A
Thin+Burn	A	A		A	C	A	A	B

^aStatistically significant differences ($\alpha = 0.10$) between 2002 and 2016 values. ↑ = values increased, ↓ = values decreased, blank = no change

^bStatistically significant effect ($\alpha = 0.10$) of beetle outbreak on treatment effect over control. ↔ = treatment and control differences inflated, →← = treatment and control differences diminished, blank = no effect

^cStatistically significant pairwise differences ($\alpha = 0.05$) at 2016 measurement. Shared letters indicate no difference. A = lowest value, blank = no difference

potential surface fire behavior where the canopy-based metrics do not. Silviculture and MPB effects had consistent influence on the thinning versus control contrast for this metric because silvicultural thinning reduces PT but MPB-caused tree mortality increases it by compounding ladder fuels with heavy surface loading, inflating the difference (in fuel loading and longevity) between thinned and unthinned units.

We found no fuel or crown fire potential differences (treatment or MPB caused) between burn-only and control treatments. This is likely because the prescribed burning treatment was mostly kept to low fire intensities to limit overstory mortality. Fire effects were also limited because many trees, including successional species, had grown to fire-tolerant sizes in the fire-excluded twentieth century. The burning effect may have been muddled by MPB if the treatment was more severe and had more strongly influenced successful beetle attacks (Wallin *et al.* 2003) or reduced overstory density as much as thinning treatments. When combined with thinning in the

form of the thin+burn treatment, however, we did observe a minor effect of MPB on burned treatments: MPB effects in the thin+burn treatments were always slightly less than in the thin-only treatment, even to the point of being absent in the CI contrast. This is because treatment more poorly differentiated MPB-caused tree mortality between thin+burn and control than between thin-only and control (leg *a* of Fig. 1). Thus, the MPB outbreak obfuscated the thin-only versus control contrast on crown fire spread (CI), but did not impact the thin+burn contrast, suggesting that differences were mostly due to prescribed burning treatment.

Combined effects: the state of treatment in 2016

Disentangling the silviculture and MPB effects on fuel treatments in 2016 is useful for understanding the relative importance of these factors on the fuel development process and on change in crown fire potential, but land managers may be more concerned with the resultant state of treatment fuel loading and crown fire potential. In this sense, treatment effectiveness at maintaining low crown fire potential may be a more practically important matter than effect mediation.

We found that thinned units had less fuel and lower crown fire potential than unthinned units in 2016. One major difference between 2002 and 2016 thinning effects was the radical increase of surface fuel (FWD and CWD) in unthinned units, which likely would not have happened without the MPB outbreak. These surface fuels are directly tied to increased surface fire behavior and potential for torching. Although thinning was a statistically significant predictor of PT in both 2002 and 2016, 2016 probabilities were higher in unthinned (50%) than thinned units and therefore more significant, and undesirable, in practical terms. This condition is typical of unmanaged second-growth ponderosa pine–Douglas fir forests impacted by beetles throughout the Interior West and reflects that torching and crowning fire behavior may be commonplace in many unmaintained, post-outbreak units (Jain *et al.* 2012).

We also found that burned units had less canopy fuel and lower probabilities of sustaining crown fire than unburned units in 2016. Interestingly, we observed delayed effects of burning on canopy fuel characteristics and crown fire potential (CBH, CBD, CI) that were not present in 2002. This delay is likely due to secondary fire-induced mortality, namely pre-MPB-outbreak attacks by other beetles on trees weakened by prescribed fire, as documented by Six and Skov (2009). Despite these two beetle episodes (endemic then epidemic) in the burn-only treatment, and that the control was only slightly different from the burn-only in all of the 2016 fuel and crown fire potential facets, FFE-FVS predicted “surface” type fire only 40% of the time in the control versus 70% in the burn-only. Fire (and fire modeling) is very sensitive to thresholds in fuels, weather, and topography. Although the

potential fire behavior metrics that this study presents are more valuable for comparative analysis than absolute characterization, they illustrate that prescribed fire may only have mild effects on measured vegetation and fuels structure, but still reduce potential fire behavior below important crowning thresholds.

The combination of thinning and burning is clearly the most effective at sustaining low crown fire potential in light of post-treatment growth and subsequent disturbance. The 2016 thin+burn was superior to the burn-only and thin-only treatments for three main reasons. First, it reduced surface and canopy fuels. Combined thinning and burning most effectively reduces fuels because thinning removes substantial tree and canopy biomass, while burning consumes surface fuels that have built up prior to thinning in addition to activity fuels (Stephens and Moghaddas 2005). Second, thin+burn reduced MPB-caused mortality relative to control and burn-only. Although thinning during outbreak-level disturbances may be ineffective (Six *et al.* 2014), thinning prior to an outbreak has been shown to moderate mortality (Fettig *et al.* 2014, Jenkins *et al.* 2014, Hood *et al.* 2016). Third, the thin+burn dampened development of ladder fuels by killing regeneration and potential ingrowth present after commercial thinning. Keyes and O'Hara (2002) identified that fuel treatments stimulate forest regeneration, and in turn negate fuel reduction objectives. Although thin+burn units are in fact regenerating, the combination of these treatments killed advanced regeneration and reset an understory development phase, lengthening the duration of treatment effectiveness. The recent divergence from thin-only emphasizes that treating the understory is imperative for extending the duration of treatment longevity. Although stand densities in both thin-only and thin+burn were most similar to stand densities in local historical units with intact frequent fire regimes (*e.g.*, per Clyatt *et al.* 2016), without re-entry or burning, the thin-only treatment may not be able to resist crown fire like the thin+burn or historical, open ponderosa pine units (Arno *et al.* 2008). Although the thin+burn treatment had slightly greater MPB-caused mortality than thin-only, fuel treatments in ponderosa pine forest types that include both thinning and burning (*i.e.*, because burning treated the understory) best establish forest structure that is able to resist both beetles and crown fire well into the second decade. This timeframe is especially important for ponderosa pine forests in the inland Northwest, where fire return intervals may range up to half a century (Arno *et al.* 1995) and managers may be financially or logistically unable to keep up fuel treatments.

Conclusion

Fuel treatments have been widely implemented to reduce crown fire potential in fire-prone forests. However, recent bark beetle outbreaks have impacted millions of hectares of

unmanaged and managed forests throughout the West. This study shows that fuel treatments followed by an MPB outbreak generate their own unique responses that differ from original treatment responses. Overall, thinned then MPB-attacked units had less fuel and lower crown fire potential than unthinned attacked units. Burned then beetle-attacked units had less canopy fuel and also had lower crown fire potential than unburned attacked units. Combined thinning and burning best improved fuel treatment longevity; even after MPB outbreak, this treatment exhibited little change in fuel profile and crown fire potential.

Bark beetle outbreaks reduce live stem densities and canopy fuels. However, bark beetle outbreaks and fuel treatments typically target different tree sizes and species, resulting in different forest structures and species composition, and these differences can have profound impacts on potential fire behavior. The MPB outbreak in our study had a complex effect on fuels and crown fire potential in treated versus untreated units, amplifying some differences and reducing others. High levels of MPB-caused mortality in control units and ladder fuel ingrowth in thin-only units made fuel and crown fire potential in these two treatments more similar in a number of ways. Despite becoming more similar to thinned treatments following MPB outbreak, control treatments were still characterized by greater crown fire potential, emphasizing a key difference between silvicultural and MPB "treatment."

Acknowledgements

This was a study of the Applied Forest Management Program at the University of Montana, a research and demonstration unit of the Montana Forest and Conservation Experiment Station. We are also indebted to A. Larson for providing feedback on an earlier draft.

Funding

This work was supported by the USDA National Institute of Food and Agriculture, McIntire Stennis project 233356 (awarded to C. Keyes). The study was made possible with the foresight of past scientists who designed and implemented the Fire and Fire Surrogate Study, a nation-wide project funded by the Joint Fire Sciences Program (FFS 99-S-01).

Availability of data and materials

Data available upon request.

Authors' contributions

JSC and CRK designed the study and acquired the data. JSC performed the analysis and wrote the manuscript. JSC, CRK, SMH, DLRA, and AS contributed to data interpretation and manuscript revision. All authors read and approved the final manuscript.

Ethics approval and consent to participate

Not applicable.

Consent for publication

Not applicable.

Competing interests

The authors declare that they have no competing interests.

Publisher's Note

Springer Nature remains neutral with regard to jurisdictional claims in published maps and institutional affiliations.

Author details

¹University of Montana, WA Franke College of Forestry and Conservation, 32 Campus Drive, Missoula, Montana 59812, USA. ²Present address: USDA Forest Service, Pacific Northwest Research Station, 11175 Auke Lake Way, Juneau, Alaska 99801, USA. ³USDA Forest Service, Rocky Mountain Research Station, Fire, Fuel, and Smoke Science Program, 5775 Highway 10 W, Missoula, Montana 59808, USA. ⁴University of Montana, Division of Biological Sciences, 32 Campus Drive, Missoula, Montana 59812, USA.

Received: 20 September 2018 Accepted: 24 September 2018

Published online: 27 December 2018

References

- Affleck, D., C. Keyes, and J. Goodburn. 2012. Conifer crown fuel modeling: Current limits and potential for improvement. *Western Journal of Applied Forestry* 27 (4): 165–169 <https://doi.org/10.5849/wjaf.11-039>.
- Agee, J.K., and C.N. Skinner. 2005. Basic principles of forest fuel reduction treatments. *Forest Ecology and Management* 211 (1): 83–96 <https://doi.org/10.1016/j.foreco.2005.01.034>.
- Albini, F.A. 1976. Estimating wildfire behavior and effects. In *USDA Forest Service general technical report INT-30*. Intermountain Forest and Range Experiment Station, Ogden, Utah: USA.
- Arno, S.F., C.E. Fiedler, and M.K. Arno. 2008. *Giant pines and grassy glades. The historic ponderosa pine ecosystem, disappearing icon of the American west, Forest history today (spring)*, 12–19.
- Arno, S.F., J.H. Scott, and G. Hartwell. 1995. Age-class structure of old growth ponderosa pine/ Douglas-fir stands and its relationship to fire history. In *USDA Forest Service research paper INT-RP-481*. Intermountain Research Station, Ogden, Utah: USA.
- Baron, R., and D. Kenny. 1986. The moderator-mediator variable distinction in social psychological research. *Journal of Personality and Social Psychology* 51 (6): 1173–1182 <https://doi.org/10.1037/0022-3514.51.6.1173>.
- Bebi, P., D. Kulakowski, and T.T. Veblen. 2003. Interactions between fire and spruce beetles in a subalpine Rocky Mountain forest landscape. *Ecology* 84 (2): 362–371 [https://doi.org/10.1890/0012-9658\(2003\)084\[0362:IBFASB\]2.0.CO;2](https://doi.org/10.1890/0012-9658(2003)084[0362:IBFASB]2.0.CO;2).
- Bentz, B.J., J. Regniere, C.J. Fettig, M. Hansen, J.L. Hayes, J.A. Hicke, R.G. Kelsey, J.F. Negrón, and S. Seybold. 2010. Climate change and bark beetles of the western United States and Canada: Direct and indirect effects. *BioScience* 60 (8): 602–613 <https://doi.org/10.1525/bio.2010.60.8.6>.
- Bigler, C., D. Kulakowski, and T.T. Veblen. 2005. Multiple disturbance interactions and drought influence fire severity in Rocky Mountain subalpine forests. *Ecology* 86 (11): 3018–3029 <https://doi.org/10.1890/05-0011>.
- British Columbia Ministry of Forests. 2004. *Bark beetle management guidebook*. British Columbia Ministry of Forests. British Columbia, Canada: Victoria.
- Brown, J.K. 1974. Handbook for inventorying downed woody material. In *USDA Forest Service general technical report INT-GTR-16*. Intermountain Research Station, Ogden, Utah: USA.
- Brown, R.T., J.K. Agee, and J.F. Franklin. 2004. Forest restoration and fire: Principles in the context of place. *Conservation Biology* 18 (4): 903–912 https://doi.org/10.1111/j.1523-1739.2004.521_1.x.
- Clyatt, K.A., J.S. Crotteau, M.S. Schaedel, H.L. Wiggins, H. Kelley, D.J. Churchill, and A.J. Larson. 2016. Historical spatial patterns and contemporary tree mortality in dry mixed-conifer forests. *Forest Ecology and Management* 361: 23–37 <https://doi.org/10.1016/j.foreco.2015.10.049>.
- R Core Team. 2016. R: A language and environment for statistical computing. <<http://www.r-project.org/>>. Accessed 9 Oct 2018.
- Covington, W.W., and M.M. Moore. 1994. Postsettlement changes in natural fire regimes and forest structure. *Journal of Sustainable Forestry* 2 (1): 153–181 https://doi.org/10.1300/J091v02n01_07.
- Cruz, M.G., and M.E. Alexander. 2010. Assessing crown fire potential in coniferous forests of western North America: A critique of current approaches and recent simulation studies. *International Journal of Wildland Fire* 19: 377–398 <https://doi.org/10.1071/WF08132>.
- Cruz, M.G., M.E. Alexander, and R.H. Wakimoto. 2005. Development and testing of models for predicting crown fire rate of spread in conifer forest stands. *Canadian Journal of Forest Research* 35 (7): 1626–1639 <https://doi.org/10.1139/x05-085>.
- Dixon, G. 2002. *Essential FVS: A user's guide to the Forest vegetation simulator*. USDA Forest Service internal report. Fort Collins, Colorado, USA: Forest Management Service Center.
- Donato, D.C., B.J. Harvey, W.H. Romme, M. Simard, and M.G. Turner. 2013. Bark beetle effects on fuel profiles across a range of stand structures in Douglas-fir forests of greater Yellowstone. *Ecological Applications* 23 (1): 3–20 <https://doi.org/10.1890/12-0772.1>.
- Fernandes, P.M. 2009. Examining fuel treatment longevity through experimental and simulated surface fire behaviour: A maritime pine case study. *Canadian Journal of Forest Research* 39 (12): 2529–2535 <https://doi.org/10.1139/X09-145>.
- Fettig, C., R. Borys, and C. Dabney. 2010. Effects of fire and fire surrogate treatments on bark beetle-caused tree mortality in the southern cascades, California. *Forest Science* 56 (1): 60–73.
- Fettig, C.J., K.E. Gibson, A.S. Munson, and J.F. Negrón. 2014. A comment on "management for mountain pine beetle outbreak suppression: Does relevant science support current policy?". *Forests* 5 (4): 822–826 <https://doi.org/10.3390/f5040822>.
- Fiedler, C.E., K.L. Metlen, and E.K. Dodson. 2010. Restoration treatment effects on stand structure, tree growth, and fire hazard in a ponderosa pine/Douglas-fir forest in Montana. *Forest Science* 56 (1): 18–31.
- Finney, M.A., C.W. McHugh, and I.C. Grenfell. 2005. Stand- and landscape-level effects of prescribed burning on two Arizona wildfires. *Canadian Journal of Forest Research* 35 (7): 1714–1722 <https://doi.org/10.1139/x05-090>.
- Flannigan, M.D., M.A. Krawchuk, W.J. de Groot, M.B. Wotton, and L.M. Gowman. 2009. Implications of changing climate for global wildland fire. *International Journal of Wildland Fire* 18 (5): 483–507 <https://doi.org/10.1071/WF08187>.
- Fulé, P.Z., J.E. Crouse, J.P. Roccaforte, and E.L. Kalies. 2012. Do thinning and/or burning treatments in western USA ponderosa or Jeffrey pine-dominated forests help restore natural fire behavior? *Forest Ecology and Management* 269: 68–81.
- Gannon, A., and S. Sontag. 2010. Montana forest insect and disease conditions and program highlights-2010. In *USDA Forest Service region 1, Forest health and protection report 11-1*. Missoula, Montana: USA.
- Grissino-Mayer, H.D., C.M. Gentry, S.Q. Croy, J. Hiatt, B. Osborne, A. Stan, and G. DeWeese Wight. 2006. *Fire history of western Montana forested landscapes via tree-ring analyses*. Indiana State University Department of geography, geology, and anthropology. Terre Haute, Indiana, USA: Professional Paper No. 23.
- Harrington, M.G., E. Noonan-Wright, M. Doherty. 2007. Testing the modeled effectiveness of an operational fuel reduction treatment in a small western Montana interface landscape using two spatial scales. Pages 301–314 in: B.W. Butler, and W. Cook, The fire environment—innovations, management, and policy. USDA Forest Service Proceedings RMRS-P-46CD, Rocky Mountain Research Station, Fort Collins, Colorado, USA.
- Hart, S.J., T. Schoennagel, T.T. Veblen, and T.B. Chapman. 2015. Area burned in the western United States is unaffected by recent mountain pine beetle outbreaks. *Proceedings of the National Academy of Sciences of the United States of America* 112 (14): 4375–4380 <https://doi.org/10.1073/pnas.1424037112>.
- Harvey, B.J., D.C. Donato, W.H. Romme, and M.G. Turner. 2013. Influence of recent bark beetle outbreak on fire severity and postfire tree regeneration in montane Douglas-fir forests. *Ecology* 94 (11): 2475–2486 <https://doi.org/10.1890/13-0188.1>.
- Hessburg, P.F., D.J. Churchill, A.J. Larson, R.D. Haugo, C. Miller, T.A. Spies, M.P. North, N.A. Povak, R.T. Belote, P.H. Singleton, W.L. Gaines, R.E. Keane, G.H. Aplet, S.L. Stephens, P. Morgan, P.A. Bisson, B.E. Rieman, R.B. Salter, and G.H. Reeves. 2015. Restoring fire-prone inland Pacific landscapes: Seven core principles. *Landscape Ecology* 30: 1805–1835 <https://doi.org/10.1007/s10980-015-0218-0>.
- Hicke, J.A., M.C. Johnson, L.H.D. Jane, and H.K. Preisler. 2012. Effects of bark beetle-caused tree mortality on wildfire. *Forest Ecology and Management* 271: 81–90.
- Hicke, J.A., A.J.H. Meddens, and C.A. Kolden. 2016. Recent tree mortality in the western United States from bark beetles and forest fires. *Forest Science* 62 (2): 141–153 <https://doi.org/10.5849/forsci.15-086>.
- Hood, S.M., S. Baker, and A. Sala. 2016. Fortifying the forest: Thinning and burning increase resistance to a bark beetle outbreak and promote forest resilience. *Ecological Applications* 26 (7): 1984–2000 <https://doi.org/10.1002/eap.1363>.
- Jain, T.B., M.A. Battaglia, H.-S. Han, R.T. Graham, C.R. Keyes, J.S. Fried, and J.E. Sandquist. 2012. A comprehensive guide to fuel management practices for dry mixed conifer forests in the northwestern United States. In *USDA Forest Service general technical report RMRS-GTR-292*. Rocky Mountain Research Station, Fort Collins, Colorado: USA.
- Jenkins, M.J., E. Hebertson, W. Page, and C.A. Jorgensen. 2008. Bark beetles, fuels, fires and implications for forest management in the intermountain west. *Forest Ecology and Management* 254 (1): 16–34 <https://doi.org/10.1016/j.foreco.2007.09.045>.

- Jenkins, M.J., W.G. Page, E.G. Hebertson, and M.E. Alexander. 2012. Fuels and fire behavior dynamics in bark beetle-attacked forests in western North America and implications for fire management. *Forest Ecology and Management* 275: 23–34 <https://doi.org/10.1016/j.foreco.2012.02.036>.
- Jenkins, M.J., J.B. Runyon, C.J. Fettig, W.G. Page, and B.J. Bentz. 2014. Interactions among the mountain pine beetle, fires, and fuels. *Forest Science* 60 (3): 489–501 <https://doi.org/10.5849/forsci.13-017>.
- Johnson, M.C., M.C. Kennedy, and D.L. Peterson. 2011. Simulating fuel treatment effects in dry forests of the western United States: Testing the principles of a fire-safe forest. *Canadian Journal of Forest Research* 41 (5): 1018–1030 <https://doi.org/10.1139/x11-032>.
- Kane, J.M., J.M. Varner, M.R. Metz, and P.J. van Mantgem. 2017. Characterizing fire-disturbance interactions and their potential impacts on tree mortality in western US forests. *Forest Ecology and Management* 405: 188–199 <https://doi.org/10.1016/j.foreco.2017.09.037>.
- Keeling, E.G., A. Sala, and T.H. DeLuca. 2006. Effects of fire exclusion on forest structure and composition in unlogged ponderosa pine/Douglas-fir forests. *Forest Ecology and Management* 237: 418–428 <https://doi.org/10.1016/j.foreco.2006.09.064>.
- Keyes, C.R., and K.L. O'Hara. 2002. Quantifying stand targets for silvicultural prevention of crown fires. *Western Journal of Applied Forestry* 17 (2): 101–109.
- Keyes, C.R., and J.M. Varner. 2006. Pitfalls in the silvicultural treatment of canopy fuels. *Fire Manage Today* 66: 46–50.
- Kolb, T.E., C.J. Fettig, M.P. Ayres, B.J. Bentz, J.A. Hicke, R. Mathiasen, J.E. Stewart, and A.S. Weed. 2016. Observed and anticipated impacts of drought on forest insects and diseases in the United States. *Forest Ecology and Management* 380: 321–334 <https://doi.org/10.1016/j.foreco.2016.04.051>.
- Larsson, S., R. Oren, R.H. Waring, and J.W. Barrett. 1983. Attacks of mountain pine beetle as related to tree vigor of ponderosa pine. *Forest Science* 29 (2): 395–402.
- MacKinnon, D.P., A.J. Fairchild, and M.S. Fritz. 2007. Mediation analysis. *Annual Review of Psychology* 58: 593–614 <https://doi.org/10.1146/annurev.psych.58.110405.085542>.
- Martinson, E.J., and P.N. Omi. 2013. *Fuel treatments and fire severity: A meta-analysis*. Fort Collins, Colorado, USA: USDA Forest Service Research Paper RMRS-RP-103WWW, Rocky Mountain Research Station <https://doi.org/10.2737/RMRS-RP-103>.
- McIver, J.D., S.L. Stephens, J.K. Agee, J. Barbour, R.E.J. Boerner, C.B. Edminster, K.L. Erickson, K.L. Farris, C.J. Fettig, C.E. Fiedler, S. Haase, S.C. Hart, J.E. Keeley, E.E. Knapp, J.F. Lehmkuhl, J.J. Moghaddas, W. Otrrosina, K.W. Outcalt, D.W. Schwillk, C.N. Skinner, T.A. Waldrop, C.P. Weatherspoon, D.A. Yaussy, A. Youngblood, and S. Zack. 2012. Ecological effects of alternative fuel-reduction treatments: Highlights of the National Fire and fire surrogate study (FFS). *International Journal of Wildland Fire* 22: 63–82 <https://doi.org/10.1071/WF11130>.
- McIver, J.D., and C.P. Weatherspoon. 2010. On conducting a multisite, multidisciplinary forestry research project: Lessons from the national fire and fire surrogate study. *Forest Science* 56 (1): 4–17.
- Metlen, K.L., and C.E. Fiedler. 2006. Restoration treatment effects on the understory of ponderosa pine/Douglas-fir forests in western Montana, USA. *Forest Ecology and Management* 222 (1): 355–369 <https://doi.org/10.1016/j.foreco.2005.10.037>.
- Miller, J.D., H.D. Safford, M. Crimmins, and A.E. Thode. 2009. Quantitative evidence for increasing forest fire severity in the Sierra Nevada and southern Cascade Mountains, California and Nevada, USA. *Ecosystems* 12 (1): 16–32.
- Moran, C.J., and M.A. Cochrane. 2012. Do mountain pine beetle outbreaks change the probability of active crown fire in lodgepole pine forests? Comment. *Ecology* 93 (4): 939–941.
- Negrón, J.F., and J.B. Popp. 2004. Probability of ponderosa pine infestation by mountain pine beetle in the Colorado front range. *Forest Ecology and Management* 191 (1–3): 17–27 <https://doi.org/10.1016/j.foreco.2003.10.026>.
- Nelson, K.N., M.G. Turner, W.H. Romme, and D.B. Tinker. 2017. Simulated fire behaviour in young, postfire lodgepole pine forests. *International Journal of Wildland Fire* 26 (10): 852–865 <https://doi.org/10.1071/WF16226>.
- Nimlos TJ (1986). Soils of Lubrecht experimental Forest. Montana Forest and conservation Experiment Station, miscellaneous publication no. 44, Missoula, Montana, USA.
- Noonan-Wright, E.K., N.M. Vaillant, and A.L. Reiner. 2014. The effectiveness and limitations of fuel modeling using the fire and fuels extension to the Forest vegetation simulator. *Forest Science* 60 (2): 231–240 <https://doi.org/10.5849/forsci.12-062>.
- Page, W.G., and M.J. Jenkins. 2007. Mountain pine beetle-induced changes to selected lodgepole pine fuel complexes within the intermountain region. *Forest Science* 53 (4): 507–518.
- Parsons, D.J., and S.H. DeBenedetti. 1979. Impact of fire suppression on a mixed-conifer forest. *Forest Ecology and Management* 2: 21–33 [https://doi.org/10.1016/0378-1127\(79\)90034-3](https://doi.org/10.1016/0378-1127(79)90034-3).
- Pfister, R.D., B.L. Kovalchik, S.F. Arno, and R.C. Presby. 1977. Forest habitat types of Montana. In *USDA Forest Service general technical report INT-GTR-34*. Intermountain Research Station, Ogden, Utah: USA.
- Pinheiro J, Bates D, DebRoy S, R Core Team. 2016. Nlme: Linear and nonlinear mixed effects models. <http://cran.r-project.org/package=nlme>. Accessed 9 Oct 2018.
- Raffa, K.F., B.H. Aukema, B.J. Bentz, A.L. Carroll, J.A. Hicke, M.G. Turner, and W.H. Romme. 2008. Cross-scale drivers of natural disturbances prone to anthropogenic amplification: The dynamics of bark beetle eruptions. *Bioscience* 58 (6): 501–517 <https://doi.org/10.1641/B580607>.
- Rebain, S. 2010. *The fire and fuels extension to the Forest vegetation simulator: Updated model documentation*. Fort Collins Available online at: USDA Forest Service, Forest Management Service Center <http://www.fs.fed.us/fmrc/ftp/fvs/docs/gtr/FFEGuide.pdf>.
- Reineke, L.H. 1933. Perfecting a stand-density index for even-aged forests. *Journal of Agricultural Research* 46: 627–638.
- Reinhardt, E.D., R.E. Keane, D.E. Calkin, and J.D. Cohen. 2008. Objectives and considerations for wildland fuel treatment in forested ecosystems of the interior western United States. *Forest Ecology and Management* 256 (12): 1997–2006 <https://doi.org/10.1016/j.foreco.2008.09.016>.
- Savage, M., and J. Nystrom Mast. 2005. How resilient are southwestern ponderosa pine forests after crown fires? *Canadian Journal of Forest Research* 35 (4): 967–977 <https://doi.org/10.1139/x05-028>.
- Schoennagel, T., T.T. Veblen, J.F. Negron, and J.M. Smith. 2012. Effects of mountain pine beetle on fuels and expected fire behavior in lodgepole pine forests, Colorado, USA. *PLoS One* 7 (1): 1–14 <https://doi.org/10.1371/journal.pone.0030002>.
- Scott, J.H.E.D. Reinhardt. 2001. Assessing crown fire potential by linking models of surface and crown fire behavior. In *USDA Forest Service research paper RMRS-RP-29*. Rocky Mountain Research Station, Fort Collins, Colorado: USA.
- Sieg, C.H., R.R. Linn, F. Pimont, C.M. Hoffman, J.D. McMillin, J. Winterkamp, and L.S. Baggett. 2017. Fires following bark beetles: Factors controlling severity and disturbance interactions in ponderosa pine. *Fire Ecology* 13 (3): 1–23 <https://doi.org/10.4996/fireecology.130300123>.
- Simard, M., W.H. Romme, J.M. Griffin, and M.G. Turner. 2011. Do mountain pine beetle outbreaks change the probability of active crown fire in lodgepole pine forests? *Ecological Monographs* 81 (1): 3–24 <https://doi.org/10.1890/10-1176.1>.
- Six, D.L., E. Biber, and E. Long. 2014. Management for mountain pine beetle outbreak suppression: Does relevant science support current policy? *Forests* 5 (1): 103–133 <https://doi.org/10.3390/f5010103>.
- Six, D.L., and K. Skov. 2009. Response of bark beetles and their natural enemies to fire and fire surrogate treatments in mixed-conifer forests in western Montana. *Forest Ecology and Management* 258: 761–772 <https://doi.org/10.1016/j.foreco.2009.05.016>.
- Stephens, S.L., B.M. Collins, and G. Roller. 2012. Fuel treatment longevity in a Sierra Nevada mixed conifer forest. *Forest Ecology and Management* 285: 204–212 <https://doi.org/10.1016/j.foreco.2012.08.030>.
- Stephens, S.L., and J.J. Moghaddas. 2005. Experimental fuel treatment impacts on forest structure, potential fire behavior, and predicted tree mortality in a California mixed conifer forest. *Forest Ecology and Management* 215: 21–36 <https://doi.org/10.1016/j.foreco.2005.03.070>.
- Stephens, S.L., J.J. Moghaddas, C.B. Edminster, C.E. Fiedler, S.M. Haase, M.G. Harrington, J.E. Keeley, E.E. Knapp, J.D. McIver, K. Metlen, C.N. Skinner, and A. Youngblood. 2009. Fire treatment effects on vegetation structure, fuels, and potential fire severity in western US forests. *Ecological Applications* 19 (2): 305–320 <https://doi.org/10.1890/07-1755.1>.
- Tinkham, W., C. Hoffman, S. Ex, M. Battaglia, and J. Saralecos. 2016. Ponderosa pine forest restoration treatment longevity: Implications of regeneration on fire hazard. *Forests* 7 (7): 137 <https://doi.org/10.3390/f7070137>.
- US Forest Service. 2005. Forest inventory and analysis National Core Field Guide Volume I: Field data collection procedures for phase 2 plots. <<https://www.fia.fs.fed.us/library/field-guides-methods-proc/>>. Accessed 9 Oct 2018.
- Wallin, K.F., T.E. Kolb, K.R. Skov, and M.R. Wagner. 2003. Effects of crown scorch on ponderosa pine resistance to bark beetles in northern Arizona. *Environmental Entomology* 32 (3): 652–661 <https://doi.org/10.1603/0046-225X-32.3.652>.
- Weatherspoon, C.P. 2000. A long-term national study of the consequences of fire and fire surrogate treatments. In *Proceedings of the joint fire science conference and workshop—Crossing the millenium: Integrating spatial technologies and ecological principles for a new age in fire management*, ed. L.F. Neuenschwander and K.C. Ryan, 117–126. Moscow: University of Idaho Press.
- Westerling, A.L., H.G. Hidalgo, D.R. Cayan, and T.W. Swetnam. 2006. Warming and earlier spring increase western US forest wildfire activity. *Science* 313: 940–943 <https://doi.org/10.1126/science.1128834>.