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Ecological effects of prescribed burning, mechanical cutting, and post-treatment wildfire for restoration of Pinus albicaulis

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ECOLOGICAL EFFECTS OF PRESCRIBED BURNING, MECHANICAL CUTTING, AND POST-TREATMENT WILDFIRE FOR RESTORATION OF *PINUS ALBICAULIS*

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ABSTRACT

The field of ecological restoration is growing rapidly, increasing the need for reliable and generalizable information on the impacts of management interventions aimed to be restorative. Prescribed burning and mechanical cutting have been proposed as primary strategies for restoration. However, there is limited information on their efficacy and effects in subalpine forest types, suggesting that monitoring to inform adaptive management is a priority need. I used data from a 15-year, replicated before-after-control-impact (BACI) study on *Pinus albicaulis* (whitebark pine) restoration to assess the ecological effects of prescribed burning and mechanical cutting, with and without subsequent unplanned wildfire, as well as the efficacy of the monitoring design. Mature tree mortality was high across all study units (77-100%), but neither treatment type nor wildfire were significant predictors of mortality. Similarly, I was unable to detect any effects of treatments or wildfire on *P. albicaulis* basal area, which declined over time across all study units. However, I found a significant effect of treatment on basal area for two (*Pinus contorta* and *Picea engelmannii)* of the three competing conifer species. At Bear Overlook, the site not affected by wildfire, *P. contorta* basal area change varied significantly between the two treatment units; it decreased by 2.1 m^2ha^{-1} in the burn-only unit but increased by 2.4 m^2ha^{-1} in the prescribed burn with mechanical cutting unit; however, neither treatment was significantly different from the control unit. In contrast, at Beaver Ridge, the prescribed burn with mechanical control

treatments, both with and without wildfire, resulted in significant reductions of *P. contorta* basal area (by 9.8 m^2ha^{-1} and 4.1 m^2ha^{-1}, respectively), compared to the untreated control (which did not experience wildfire), which increased by 1 m^2ha^{-1}. For *P. engelmannii,* at Bear Overlook, the site not affected by wildfire, basal area increased after treatment (by 10.3 m^2ha^{-1} and 2.6 m^2ha^{-1} in the burn-only and prescribed burning with mechanical cutting, respectively), but these increases did not differ from changes in the control unit (7.2 m^2ha^{-1}). *Pinus albicaulis* seedling density decreased across both sites and all treatments, however, response to treatment was not statistically significant, while response to wildfire was. The most precisely estimated variable was basal area with a 34% margin of error, followed by mortality (47%) and seedling density (71%). Overall, my findings reveal that the restoration treatments did not affect *P. albicaulis* mature tree mortality, basal area or seedling density, and were not consistently effective at reducing pressure from competing conifers 15-years after treatment. Although the study utilized best practice design (BACI) and had a relatively large number of replicates (n= 5), loss of study sites due to wildfire coupled with low precision of estimation in field measurements limited power of detection, and highlights the need for large-scale long-term monitoring networks and innovative sampling designs to improve understanding of the efficacy and effects of restoration treatments in *P. albicaulis* and other degraded forest ecosystems.

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El enfoque que creamos como practicantes de la ciencia y artífices del nuevo conocimiento es un reflejo de nuestro proceso de aprendizaje, el cual, finamente curado y refinado por la mano de nuestros mentores, interpreta desde su prisma los paradigmas de nuestro entorno. Mi proceso, a falta de referencias externas, refleja el discontinuo de mi adaptación, descubrimiento, y re-encanto con la ciencia qué experimenté durante estos accidentados años con sazón pandémico.

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1. Introduction

Rapid growth in the field of ecological restoration is increasing the need for reliable and generalizable information on the efficacy and effects of management practices. Despite widespread scientific agreement that long-term replicated Before-After-Control-Impact (BACI) designs are required to assess treatment efficacy and effects (Osenberg et al. 2006; Nelson 2021), there is still limited application of this design as well as lack of understanding about challenges with its implementation. For forest restoration, it is particularly important to understand how changing environmental conditions and ecological disturbances (e.g., wildfire regimes, droughts, pathogen outbreaks) may affect ecosystem responses to restoration practices. Specifically, there is a need to understand how stochastic events, such as wildfire, can impact treatment effects and the capacity of the sampling design to detect a response. Here, I address the ecological response to mechanical cutting and prescribed burning, two treatments that are commonly applied in coniferous forests of western North America for achieving restoration goals (Schoennagel et al. 2009; Stephens et al. 2009; Larson et al. 2012; Maher et al. 2018), and the effect of unplanned post-treatment wildfire, using a 15-year replicated BACI monitoring study on restoration treatments in *Pinus albicaulis* (whitebark pine) forests in the Rocky Mountains of western North America.

There is relatively little understanding of the ecological effects of stand management practices for restorative purposes in forest types without commercial value that experience less frequent, mixed-severity and stand-replacement fires, such as subalpine and treeline ecosystems (Arno 2001; USFS 2012). In dry, low- to mid-elevation coniferous forest that once experienced frequent, low- to moderate-intensity fire, thinning and burning treatments have been broadly implemented to reduce fire hazard and increase stand resistance to severe effects of wildfire (Schoennagel et al. 2009; Schoennagel & Nelson 2011), and there is a relatively large body of literature on treatment efficacy and effects (Omi & Joyce 2003; Nelson et al. 2008; Safford et al. 2012; Collins et al. 2014). However, more attention is needed on the efficacy and effects of treatments in other ecosystem types, like subalpine forests, and the effect of these treatments when natural wildfire events occur.

Pinus albicaulis is a species of high conservation need with limited understanding of its response to management interventions. This makes it an ideal candidate for assessing the ecological response of restoration practices (Keane & Parsons 2010a; Maher et al. 2018; Retzlaff et al. 2018). This upper subalpine tree is considered a foundational species in high-elevation forest communities of western North America (Tomback et al. 2001) due to its keystone effects on the structure, composition, and function of these ecosystems (Ellison et al. 2005). Like some other tree species (Van Mantgem et al. 2009), its populations have undergone a dramatic decline in recent decades (Smith et al. 2008). The primary causes of mortality include a native beetle, *Dendroctonus ponderosae*, and an invasive pathogen, *Cronartium ribicola* (Macfarlane et al. 2013). In addition to causing mortality, the combined effect of insect outbreaks and pathogen infections have been reported to reduce tree vigor (Jean et al. 2011) and rates of seeds and cone production (Keane & Arno 1993; Barringer et al. 2011; Shepherd et al. 2018). Furthermore, there is concern that extensive and successful fire-exclusion policies during the last century may have contributed to population declines by reducing the area burned under natural conditions in *P. albicaulis* forests, allowing for shifts in composition to shade-tolerant conifers such as *Picea engelmannii* and *Abies lasiocarpa* (Arno 1986; Keane & Arno 1993; Keane 2001; Kendall & Keane 2001), as well as reducing the abundance of non-forested patches created by mixed-severity fires in subalpine forests, which are thought to be essential for Clark's nutcracker caching habits (Tomback et al. 1990; Norment 1991), although these trends have not been well documented across the range of *P. albicaulis*. Thus, changes in disturbance regimes and forest structure could potentially affect the behavior of nutcrackers, ultimately affecting *P. albicaulis* seed dispersal and regeneration.

Concern over threats to *P. albicaulis* forests have led to its listing as an at-risk species under both the US and Canadian Endangered Species Acts (COSEWIC 2012; USFWS 2020) and the IUCN Red List (Mahalovich & Stritch 2013), and prompted management agencies to adopt coordinated, trans-boundary restoration strategies, such as the "*Range-Wide Restoration Strategy for Whitebark Pine*" (Keane et al. 2012) and the "*National Whitebark Pine Restoration Plan*" (Tomback & Sprague 2022). The range-wide strategy for *P. albicaulis* calls for mechanical removal of shade-tolerate competing species specifically to improve stand health, create fuel-bed conditions that would allow for use of prescribed burning to release *P. albicaulis* stands from competition, promote natural regeneration and create diverse age-class structures to maintain ecosystem function (Keane et al. 2012). Although there is some evidence that mechanical cuttings of shade-tolerant conifers may have beneficial effects, including increasing growth rates of *P. albicaulis* (Keane et al. 2007; Retzlaff et al. 2018), mitigating damage caused by *D. ponderosae* and *C. ribicola*, and increasing cone production (González-Ochoa et al. 2004; Lahr & Sala 2014), non-conclusive and negative responses to thinning also have been observed (Maher et al. 2018). In addition to thinning, there is interest in using other silvicultural treatments such as nutcracker opening treatments to promote regeneration by mimicking patchy and mixed-severity fires that are thought to creating openings for nutcrackers to cache seeds (Norment 1991). However, to date there is little information on the efficacy of artificial nutcracker openings for *P. albicaulis* recruitment.

In addition to mechanical harvest, the range-wide restoration strategy calls for prescribed fire to emulate wildfire regimes of *P. albicaulis* communities, release *P. albicaulis* stands from competition, recover spatial heterogeneity, and promote natural regeneration and diverse age-class structure to maintain ecosystem function (Keane et al. 2012). Although adding fire back on the landscape via prescribed fire can be restorative in some forest types (Safford et al. 2012; Stevens-Rumann et al. 2013), these fires may also increase mortality of mature *P. albicaulis* trees. Modeling approaches have shown that fire can be as much as a threat as benefit (Cary et al. 2017; Hood & Lutes 2017), and there is some field evidence that trees that experience any amount of burn damage to their boles may have high rates of mortality (Nelson & Keville 2018; Cansler et al. 2020), suggesting that more information is needed to improve the effectiveness of this range-wide recommended treatment. The wide variety of responses to prescribed burning highlights the critical importance of monitoring efforts after treatment to ensure that restoration objectives are being met (Keane 2018).

An important aspect of understanding the efficacy and effects of treatment on *P. albicaulis* stands is to understand their impacts in stands that subsequently burn by wildfire. In recent decades, the frequency, size, and severity of wildfires has increased in western U.S. forests (Westerling et al. 2006; North et al. 2012). Given that treated stands have an increased probability of burning, there is a need for information on the effects of restoration treatments in the context of wildfire (Stevens et al. 2014). Effects of thinning and burning on fire behavior in dry forests has been relatively well studied using both modelling (Schmidt et al.

2008; Vaillant et al. 2009) and field experiments (Pollet & Omi 2002; Ritchie et al. 2007; Lezberg et al. 2008; Prichard et al. 2010; Safford et al. 2012; Martinson & Omi 2013). However, treatment response to wildfire has not been similarly assessed for upper subalpine *P. albicaulis* forests, suggesting that more information is needed to improve the long-term effectiveness of recommended treatments in the range-wide restoration strategy.

Developing effective monitoring programs to understand treatment effects is challenging in general, but especially so for species that exhibit high spatial variability and complex regeneration dynamics, both of which are true for *P. albicaulis* ecosystems (Landenburger et al. 2008; Larson & Kipfmueller 2010). To make reasonable inferences from studies of the effects of management interventions, it is critical to assess the efficacy of the research or monitoring design and to adaptively change designs as necessary (Osenberg et al. 2006; Nelson 2021; Tomback et al. 2022). Maximizing efficiencies of monitoring designs is especially important given that land managers must balance generating information for decision-making with limited funds and personnel.

Currently, few mechanical cuttings and even fewer prescribed burns have been monitored for their ecological effects on *P. albicaulis* communities (Tomback et al. 2022). The available information shows contrasting results and potential study design limitations (Maher et al. 2018; Nelson & Keville 2018), suggesting that monitoring is a priority need. Additionally, there is a need for monitoring efforts to use designs that can separate treatments impacts from underlying spatial and temporal variability, which can be high for many forest structure and functionality proxy's responses, including seedling establishment (Youngblut & Luckman 2013), and that capture responses over long timeframes. Here, I took advantage of a long-term (15-year) replicated BACI study to investigate the efficacy and ecological effects of mechanical cutting and prescribed burning in *P. albicaulis* forests. After implementation of restoration treatments, several wildfires burned through some experimental units, allowing me to ask questions about both treatment effects and post-treatment responses to wildfire, as well as to explore the effectiveness of the sampling design. My specific research objectives and questions were:

(1) *Describe ecological responses to treatment:* What is the effect of treatment (prescribed burn, and prescribed burn with mechanical cutting) on *Pinus albicaulis* mortality, abundance, and regeneration, as well as the abundance of competitor conifer species (*Abies lasiocarpa*, *Picea engelmannii*, *Pinus contorta*), over a 15-year period?

(2) *Evaluate treatment responses to wildfire*: What is the impact of wildfire on the effects of treatment (prescribed burn with mechanical cutting) on *Pinus albicaulis* mortality, abundance, and regeneration, as well as the abundance of competitor conifer species over a 15-year period?

(3) *Quantify drivers of individual tree mortality:* To what extent do individual tree characteristics (height, diameter at breast height, live crown base height), site condition (pre-treatment basal area), and treatment intensity (area burned or basal area removed) affect individual tree mortality of *Pinus albicaulis* over a 15-year period?

(4) *Describe the efficacy of the monitoring design*: a) What precision of estimation was achieved in the measurements of each *Pinus albicaulis* study variable (mortality, abundance, and regeneration); and b) What level of replication is needed to achieve different levels of precision of estimation for each of these study variables?

2. Materials and Methods

This study used data from an on-going long-term monitoring project "Restoring Whitebark Pine Ecosystems" (RWPE) (Keane & Parsons 2010; Keane & Parsons 2010b), which was the first study designed to test effects of selective thinning and prescribed burnings as proactive restoration treatments in declining *P. albicaulis* forests – and remains the most comprehensive to date. The project aimed to understand the efficacy of thinning and prescribed burning at enhancing *P. albicaulis* growth and survival, killing subalpine fir without damaging associated mature *P. albicaulis* overstory, and creating caching sites for Clark's Nutcrackers and microsites suitable for *P. albicaulis* regeneration. It included a combination of experimental mechanical cuttings, prescribed burning with or without mechanical cutting, and control treatments. Treatment units were measured before and for up to 21 years after treatments.

2.1. Site Selection

The original study was implemented at five sites (Bear Overlook, Beaver Ridge, Coyote Meadows, Musgrove, and Smith Creek) located on the Bitterroot, Salmon, and Clearwater National Forests in the northern Rocky Mountains of the United States (Keane & Parsons 2010). Generally, the sites were located close to roads or trails to reduce travel time and maximize the number of plots sampled over the field season, and where there was support from the Ranger Districts for implementing the planned treatments. The majority of sites were in later stages of succession, and prior to treatment the overstory consisted of stands dominated by 200- to 400-year-old *P. albicaulis,* with associated *Abies lasiocarpa*, *Picea engelmannii*, and *Pinus contorta*. The understory was composed mostly of seedling and sapling *A. lasiocarpa* with occasional stagnated *P. albicaulis* saplings. The dominant understory plant species were primarily *Vaccinium scoparium* (grouse whortleberry), *Luzula hitchcockii* (smooth woodrush), and *Xerophyllum tenax* (bear grass). Sampling of fire scars across sites revealed a history of mixed-severity and stand replacing fires (Keane & Parsons 2010).

Table 1. Sites used for each research question, including treatments (and their replication), and post-treatment wildfire occurrence. Each treatment stand included 10 plots. Treatment codes: control = untreated; burn = prescribed burning; mec = mechanical cutting; and burn + mec = prescribed burning with mechanical cutting.

Site	Research questions	Treatments (and replication)	Wildfire
Smith Creek	4	control (1)	yes
		burn + mec (1)	yes
		mec (1)	yes
Beaver Ridge	2,4	control (1)	no
		burn + mec (1)	no
		burn + mec (1)	yes
Bear Overlook	1,3,4	control (1)	no
		burn (1)	no
		burn + mec (1)	no
Musgrove	4	control (1)	yes
		burn (1)	yes
		burn + mec (1)	yes
Coyote Meadows	4	control (3)	yes
		mec (3)	yes

For this study, I used all five of the original sites, but not all sites were used to address all research questions (table 1). To assess ecological effects of treatments (question 1), I only used data from Bear Overlook, because it was the only site with all units unaffected by post-treatment wildfire. To assess effects of wildfire on response to treatments (question 2), I only used data from Beaver Ridge, because it was the only site in which there were treated stands that had and had not experienced wildfire. To assess the influence of treatment, site conditions, and tree characteristics on individual tree mortality (question 3), I only used data from Bear Overlook, because it was the only site not affected by post-treatment wildfire and there were still live trees. Finally, to assess the efficacy of the monitoring design (question 4), I used pre-treatment data collected from untreated control units for all five sites (Bear Overlook, Beaver Ridge, Coyote Meadows, Musgrove, and Smith Creek).

2.2. Treatments

At Bear Overlook and Beaver Ridge, the sites that I used to test treatment effects and treatment response to wildfire (questions 1 and 2), three types of treatments were implemented: 1) *prescribed burning*, intended to mimic fire regimes present across *P. albicaulis* distribution, and aimed predominantly to reduce the abundance of competing conifers; 2) creation of *nutcracker openings* by mechanically cuttings all trees from competing species within a circular area of varying size (0.04-0.08 to 0.4-0.8 acres); and 3) *slashing* created by thinning all trees from competing species and leaving the slash to enhance fuelbed properties. Beaver Ridge had combinations of mechanical treatments (nutcracker openings or slashing) with prescribed fire, although the nutcracker opening with prescribed fire treatment was not planned but occurred when the nutcracker opening treatment was accidentally burned by spotting fire that spread from the adjacent prescribed burning unit during treatment implementation. Each study site also included a control (untreated) unit adjacent to the treatment units. All treatments were implemented between 1999 and 2001.

2.3. Post-treatment Wildfire

I used data from the Beaver Ridge sites to test the impact of wildfire on “prescribed burning with mechanical cutting without wildfire” and “prescribed burning with mechanical cutting and wildfire” treatments. At this site, in the summer of 2000, wildfire burned through multiple treatment units but not the untreated control unit. Data on the weather and fuel moisture during fire are not available.

2.4. Sampling Design

Within each site, 10 0.04-ha plots (macroplot) were located across the treatments units to record changes in ecological conditions. Ten plots was the maximum number that would fit within the treatment units, given their small size and irregular shape. The plots were located using a systematic approach with random start to account for the limited area and odd shape of treatment units, and concern about finding plots in later years using a random start (Keane & Parsons 2010). All plots were mapped using compass bearings and distances from benchmarks (bearing or blazed trees) and later with GPS. Trees, seedlings, and understory vegetation density, height, and cover measurements were taken prior to the treatment (Table 2), then one year after the treatment(s), and every five years after the treatment. Sampling across all sites was done within a two-to-three-week period each year, in order to have relatively consistent phenologic conditions.

2.4.1. Mature Trees & Saplings

All live mature trees (above 12 cm of diameter at breast height (DBH) and greater than 1.37 m tall) were tagged using numbered aluminum (unburned units) or stainless-steel casket tags (burn units) nailed at the center of the tree bole at DBH facing plot center. The DBH, tree height, live crown base height (LCBH), tree status (live or dead after initial tagging), and crown scorch (percentage) was recorded for each tagged tree. Saplings (trees less than 12 cm DBH and greater than 1.37 m tall) were not tagged; however, crews recorded the number of live saplings in 2.5 cm DBH size classes, and height.

2.4.2. Seedlings

Tree seedlings (trees less than 1.37 m in height) were counted by 0.5 m height classes on a 125 m^2 (0.0015 ha) circular plot (subplot) nested within the 0.04 ha plot.

Table 2. Variables measured for trees, sapling, seedlings, and treatment intensity, including sampling frame (macroplot or subplot) and measurement units. plot section. Variable codes: DBH = diameter at breast height; LCBH = Live crown base height. PIAL= *Pinus albicaulis*.

Variable		Sampling frame	Units
Trees and saplings	Species	Macroplot	-
	Status (alive or dead)	Macroplot	-
	DBH	Macroplot	cm
	LCBH (trees only)	Macroplot	m
	Height	Macroplot	m
	Crown scorch (PIAL trees only)	Macroplot	%
PIAL seedlings	Seedling height class	Subplot	m
Treatment intensity	Area burned	Macroplot	%
	Basal area removed	Macroplot	%

2.4.3. Percentage of Plot Area Burned

At each plot, US Forest Service (USFS) crews estimated the percentage of the plot area that was burned by the prescribed fire using cover classes: < 1%, 1%–5%, >5%–15%, >15%–

25%, >25%–35%, >35%–45%,>45%–55%, >55%–65%, >65%–75%, >75%–85%,>85%–95%, and >95%–100% (Lutes et al. 2006). Severity of the burn was not recorded.

2.5. Statistical Analysis

To assess ecological effects of prescribed burning and mechanical cutting on plot-level *P. albicaulis* mortality, abundance of *P. albicaulis* and competitor conifer species, and *P. albicaulis* seedling recruitment (question 1), I tested for differences among treatments (prescribed burning alone, prescribed burning with mechanical cutting, and untreated control) in *P. albicaulis* mature tree mortality, change in abundance of *P. albicaulis* and competitor conifer species, and change in *P. albicaulis* seedling density, using data from Bear Overlook. *P. albicaulis* mature tree mortality was calculated at the plot level as proportion of tagged mature live trees sampled pre-treatment that were dead 15-years post-treatment (1996-2015). Change in abundance was calculated at the plot level as raw change in basal area of all live trees (tagged and non-tagged) between pre- and 15-years post-treatment measurements for *P. albicaulis* and three competitor conifer species (*A. lasiocarpa*, *P. engelmannii*, *P. contorta*), independently. *P. albicaulis* seedling density was calculated at the plot level as raw change in number of seedlings between pre- and 15-year-post-treatment measurements.

I tested for normality in each response variable using Shapiro-Wilks tests (Shapiro & Wilk 1965) and, because of lack of normality, used non-parametric Kruskal-Wallis tests (Kruskal & Wallis 1952) to determine statistical differences between experimental units, with separate tests for each response variable (*P. albicaulis* mature tree mortality, change in abundance of *P. albicaulis* and competitor conifer species, and change in *P. albicaulis* seedling recruitment). If statistical significance was found ($p \leq 0.05$), I implemented a post-hoc Dunn's test (Dunn 1961) to determine statistical difference between paired treatments.

To assess the effects of wildfire on treatments (prescribed burning with mechanical cutting plus wildfire, prescribed burning with mechanical cutting without wildfire, and untreated and unburned control) at Beaver Ridge (question 2), I used non-parametric Kruskal-Wallis tests, with separate tests for each response variable, as described above. If statistical significance was found, I implemented a post-hoc Dunn's test to determine statistical differences between treatments.

To assess factors influencing individual tree mortality (question 3), I implemented a logistic regression (LR) using 15-year post-treatment data from Bear Overlook. Individual trees were considered as experimental units. I included individual tree mortality as the dependent variable; DBH, tree height, LCBH, and percentage of crown scorch as explanatory variables for the saturated model; and plot-level pre-treatment basal area and treatment intensity (percent area burned and relative basal area removed during treatment) as covariates. Additionally, I tested for significant interactions between DBH and basal area removed, and DBH and area burned. For this model, I included only *P. albicaulis* tagged trees that were alive pre-treatment. To calculate the odds ratio achieved by coefficients, I used the equation:

$$Odds\ ratio = e^{\beta} \tag{1}$$

Where "e" is the base of the natural logarithm, and "β" is the estimated coefficient for the variable of interest. To calculate odds as percentage, I multiplied the calculated odds ratio by 100. To test for the model goodness-of-fit, I calculated McFadden's Pseudo-R^2 (McFadden 1974).

To assess the precision of estimation in measurements of *P. albicaulis* response variables (question 4), I calculated the relative margin of error (%) around the mean achieved with the sampling design for three *P. albicaulis* variables (mature tree mortality (%), stand basal area ($m^2 ha^{-1}$), and seedling density (ind ha^{-1}), and the number of replicates required to achieve higher levels of precision. For this analysis, I used data collected during the site-establishment year for units scheduled to be left as untreated controls from all five sites, with 10 plots (subsamples) per unit.

To calculate the margin of error (ME) achieved, I used a confidence level of 95% and the equation:

$$Margin\ of\ Error\ (ME) = \sqrt{\frac{z^2 * s^2}{n}} \quad (2)$$

Where "z" is the critical value from a normal distribution (z-score), "s" is the standard deviation of the sample for the variable of interest, and "n" is the total numbers of plots. To calculate relative margin of error (RME), as percentage, I divided the calculated ME by the mean and multiplied by 100.

To calculate number of replicates (plots) needed for different levels of precision, I used the following equation, again with a confidence level of 95%.

$$Number\ of\ plots = \left(\frac{z^2 * s^2}{(ME)^2}\right) \quad (3)$$

Where "z" and "s"are as described above, and ME is the margin of error required to achieve different levels of RME, from 10% to 60% in increments of 10% (e.g., the ME used to calculate the number of plots needed to achieve a 10% RME was calculated as 0.1 multiplied by the mean). RME achieved and number of plots required was calculated separately for each response variable: mature tree mortality, stand basal area, and seedling density. If the calculated number of plots required had decimal places, the number was rounded up to keep an integer outcome.

Statistical analyses were done using R (Team 2018) and R Studio version 1.1.453 (RStudio 2016). All P-values are reported following the guidelines from the American Statistical Association (Wasserstein & Lazar 2016).

3. Results

3.1. Ecological response to treatment

At Bear Overlook, the area that did not burn by wildfire, *P. albicaulis* mature tree mortality was about 77% across all plots 15 years after treatment implementation. Seventy-four of the 102 trees that were initially tagged were dead by the last year of measurement. Mortality was 78% for the prescribed burning alone treatment (34 of 48 tagged trees died) and 80% for the prescribed burning with mechanical cutting treatment (22 of 27 tagged trees died). These rates of mortality were about 6 percentage points higher than in the untreated control stand (73% mortality; 18 of 27 tagged trees died), but differences among units were not statistically significant (H= 0.56, df= 2, p= 0.75, figure 1A, figure 2A).

Over the 15-year period, *P. albicaulis* basal area decreased within all units: from 5.0 m^2ha^{-1}(±1.2 SE) to 1.7 m^2ha^{-1} (±0.5 SE) for the prescribed burning alone unit; from 4.4 m^2ha^{-1} (±0.8 SE) to 1.6 m^2ha^{-1} (±0.6 SE) for the prescribed burning with mechanical cutting unit; and from 3.3 m^2ha^{-1} (±0.4 SE) to 1.1 m^2ha^{-1} (±0.3 SE) for the untreated control stand. However, there were not significant differences among units in basal area declines (H= 0.01, df= 2, p= 0.99, figure 1B, figure 2B). In contrast, *A. lasiocarpa* basal area increased in all units: from 0.9 m^2ha^{-1} (±0.5 SE) to 2.8 m^2ha^{-1} (±0.4 SE) for the prescribed burning alone unit; from 1.5 m^2ha^{-1} (±0.4 SE) to 2.8 m^2ha^{-1} (±0.9 SE) for the prescribed burning with mechanical cutting unit; and from 2.3 m^2ha^{-1} (±0.5 SE) to 4.2 m^2ha^{-1} (±0.6 SE) for the untreated control stand. There were not significant differences among units in basal area declines (H= 1.49, df= 2, p= 0.47). *P. engelmanii* basal area also increased in all units: from 3.6 m^2ha^{-1} (±1.0 SE) to 13.9 m^2ha^{-1} (±1.3 SE) for the prescribed burning alone unit; from 5.0 m^2ha^{-1} (±1.7 SE) to 7.6 m^2ha^{-1} (±1.5 SE) for the prescribed burning with mechanical cutting; and from 3.2 m^2ha^{-1} (±0.7 SE) to 10.4 m^2ha^{-1} (±0.9 SE) for the untreated control stand. This trend was statistically significant (H= 6.34, df= 2, p= 0.04): the burn-only stand was significantly different from the stand that was burned with mechanical cutting (p= 0.01); however, neither treated stand was significantly different from the untreated control stand (p= 0.25 and 0.17, respectively). Finally, *P. contorta* basal area decreased from 13.2 m^2ha^{-1} (±3.1 SE) to 11.2 m^2ha^{-1} (±1.9 SE) for the prescribed burning alone unit; increased from 2.9 m^2ha^{-1} (±1.2 SE) to 5.3 m^2ha^{-1} (±1.1 SE) for the prescribed burning with mechanical cutting unit; and increased from 6.1 m^2ha^{-1} (±1.7 SE) to 7.4 m^2ha^{-1} (±1.4 SE) for the untreated control stand. This trend was marginally significant (H= 5.37, df= 2, p= 0.06): the burn-only stand was significantly different from the stand that was burned with mechanical cutting (p= 0.02); however, neither treated stand was significantly different from the untreated control stand (p= 0.09 and 0.55, respectively).

Over the 15-year period, density of *P. albicaulis* seedlings decreased in all stands: from 343 ind ha^{-1} (±121 SE) to 140 ind ha^{-1} (±46 SE) for the prescribed burning only treatment; from 143 ind ha^{-1} (±34 SE) to 63 ind ha^{-1} (±18 SE) for the prescribed burning with

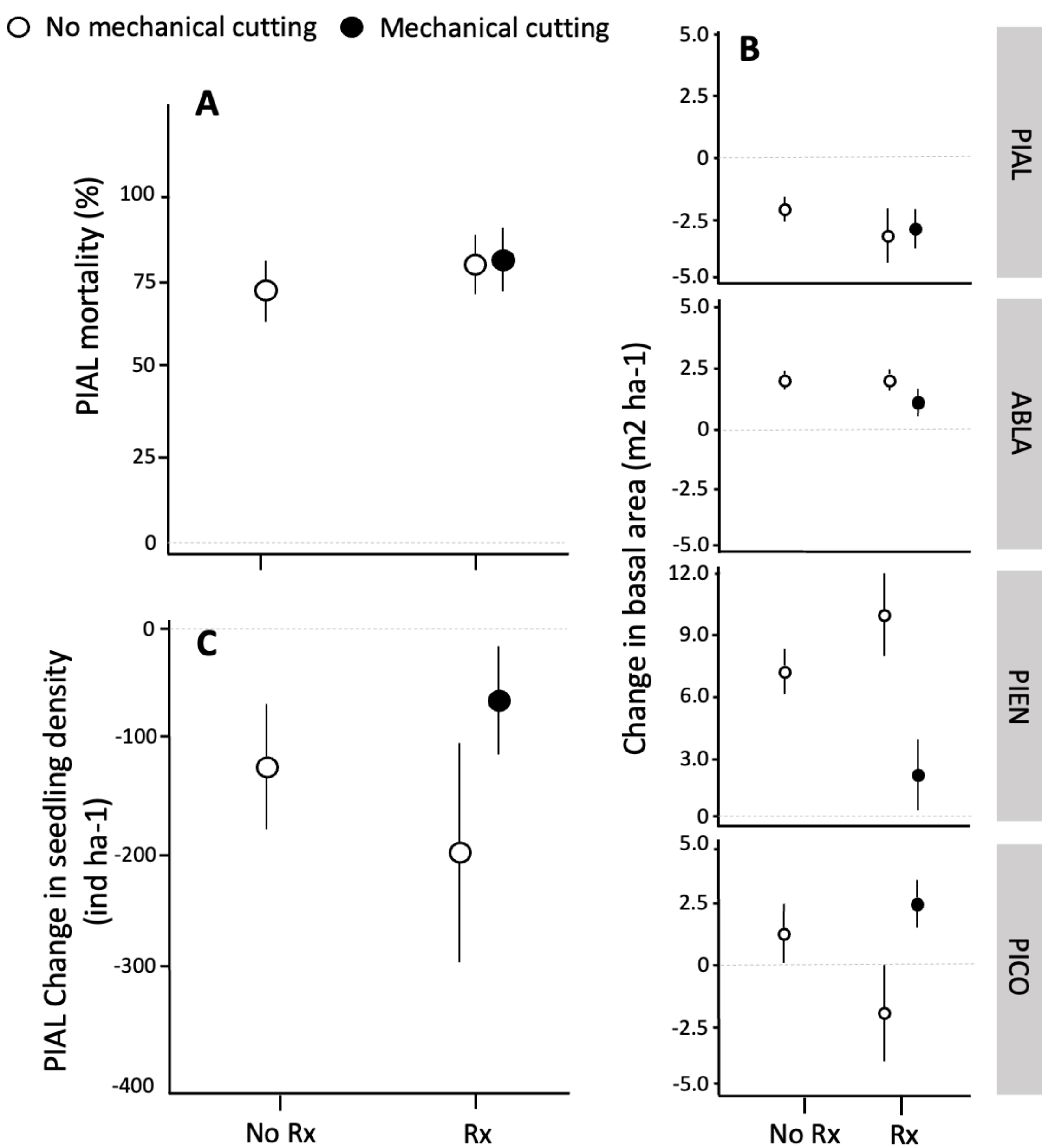

Figure 1. Effect of prescribed burning (x-axis) and mechanical cutting (solid circles, treated; open circles, not treated) on (A) mean (± 1 SE) mature *Pinus albicaulis* tree mortality (% over the 15-year period); (B) mean (± 1 SE) change (change pre to post treatment) in live tree basal area ($m^2 ha^{-1}$) of *Pinus albicaulis* and three shade-tolerant conifer species; and (C) mean (± 1 SE) change (change pre to post treatment) in *Pinus albicaulis* seedling density (ind ha^{-1}). PIAL = *Pinus albicaulis*, ABLA = *Abies lasiocarpa*, PIEN = *Picea engelmanii*, PICO = *Pinus contorta*. Data are from Bear Overlook.

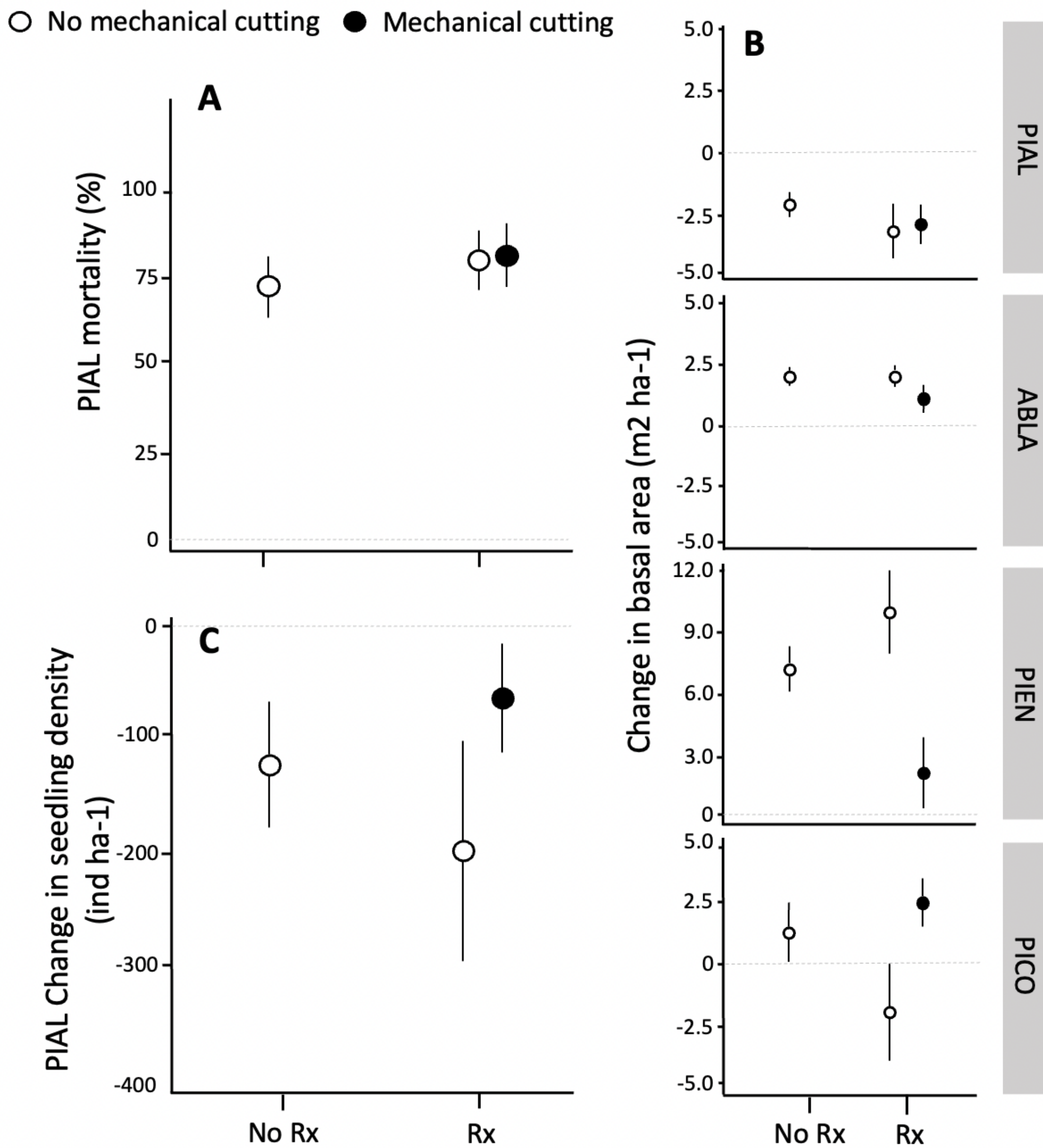

Figure 2. Effect of mechanical cutting and prescribed burning, and treatment effect (gray, pre-treatment; black, post-treatment) on (A) density of live mature *Pinus albicaulis* tree (ind ha^{-1}); (B) mean basal area change ($m^2 ha^{-1}$) of *Pinus albicaulis* and three shade-tolerant conifer species; and (C) mean *Pinus albicaulis* seedling density (ind ha^{-1}). PIAL = *Pinus albicaulis*, ABLA = *Abies lasiocarpa*, PIEN = *Picea engelmanii*, and PICO = *Pinus contorta*. C = control, T= treated with prescribed burning and mechanical cutting, Rx= treated with prescribed burning, Rx+M= treated with prescribed burning and mechanical cutting. Data are from Bear Overlook.

mechanical cutting treatment stand; and from 250 ind ha^{-1} (±71 SE) to 113 ind ha^{-1} (±35 SE) for the untreated control stand. This trend was not statistically significant (H= 0.68, df= 2, p= 0.71) (figure 1C, figure 2C).

Table 3. Summary statistics (mean (SE)) for pre-treatment total basal area for all species (Pre-Trt BA); 15-year post-treatment basal area for all species (15YR Post-Trt BA); competing conifers basal area removed (BAR); and plot area burned during treatment (Area Burned). Treatment code: control = untreated unit, burn = prescribed burning, burn + mec = prescribed burning with mechanical cutting, burn + mec + wildfire = prescribed burning with mechanical cutting and wildfire.

Site / *Treatment*	**n**	**Pre-Trt BA** *($m^2\ ha^{-1}$)*	**15YR Post-Trt BA** *($m^2\ ha^{-1}$)*	**BAR** *(%)*	**Area Burned** *(%)*
Bear Overlook					
control	10	24 (2.8)	17 (1.5)	-	-
burn	10	29 (1.9)	17 (1.5)	-	16 (3.5)
burn + mec	10	21 (1.8)	11 (1.7)	28 (13.5)	23 (6.9)
Beaver Ridge					
control	10	21 (1.9)	21 (3.4)	-	-
burn + mec	10	15 (2.6)	3 (1.1)	10 (3.6)	33 (2.6)
burn + mec + wildfire	10	10 (2.0)	2 (0.8)	46 (12.2)	84 (4.5)

3.2.Treatment response to wildfire

Pre-treatment, 11 of 30 plots across the three stands at Beaver Ridge had mature live trees. However, none of the 19 tagged trees within these 11 plots were alive by the 15th year of measurement (figure 3A, figure 4A).

Over the 15-year period, *P. albicaulis* basal area decreased within all units: from 1.8 m^2ha^{-1} (±0.9 SE) to 0.7 m^2ha^{-1} (±0.7 SE) for the treated stand not affected by wildfire (prescribed burning and mechanical cutting); from 1.1 m^2ha^{-1} (±0.8 SE) to 0.0 m^2ha^{-1} for the treated stand affected by wildfire (prescribed burning with mechanical cutting and wildfire); and from 3.2 m^2ha^{-1} (±1.0 SE) to 0.2 m^2ha^{-1} (±0.1 SE) for the untreated control stand. However, this trend was not statistically significant (*H*= 0.93, df= 2, *p*= 0.62, figure 3B, figure 4B). Basal area of *A. lasiocarpa* increased in the two units where it was present pretreatment: from 0.0 m^2ha^{-1} to 2.0 m^2ha^{-1} (±0.6 SE) for the prescribed burning with mechanical cutting treatment not affected by wildfire; and from 4.3 m^2ha^{-1} (±1.0 SE) to 6.4 m^2ha^{-1} (±1.5 SE) for the untreated control stand. This trend was not statistically significant (*H*= 0.18, df= 2, *p*= 0.67). There were not live *A. lasiocarpa* trees in the treated stand affected wildfire, either before or after treatment. Basal area of *P. engelmanii* increased in both units where it was present: from 0.0 $m^2ha^{-1}$to 0.3 m^2ha^{-1} for the only plot where it was present in the prescribed burning with mechanical cutting treatment not affected by wildfire; and from 1.4 m^2ha^{-1} (±0.7 SE) to 1.7 m^2ha^{-1} (±1.7 SE) for the untreated control stand. There were not statistically significant differences between the magnitude of increase in these units (*H*= 1.51, df= 2, *p*= 0.22). *P. engelmanii* was not present in the prescribed burning with mechanical cutting and wildfire stand either before or after treatment. Finally, basal area of *P. contorta* decreased in all treatment units: from 10.3 m^2ha^{-1} (±2.6 SE) to 0.5 m^2ha^{-1} (±0.2 SE) for the prescribed burning with mechanical cutting without wildfire unit; from 4.8 m^2ha^{-1} (±1.4 SE) to 0.8 m^2ha^{-1} (±0.4 SE) for the prescribed burning with mechanical cutting with wildfire unit; and from 2.3 m^2ha^{-1} (±0.7 SE) to 3.3 m^2ha^{-1} (±1.2 SE) for the untreated control stand. There was a statistically significant difference among units (*H*= 16.00, df= 2, *p*> 0.001):both treated stands affected and not affected by wildfire varied significantly from the untreated control stand (*p*> 0.001 and 0.02 respectively); however, change in *P. contorta* basal area did not vary significantly between the treated stands with and without wildfire (*p*= 0.26).

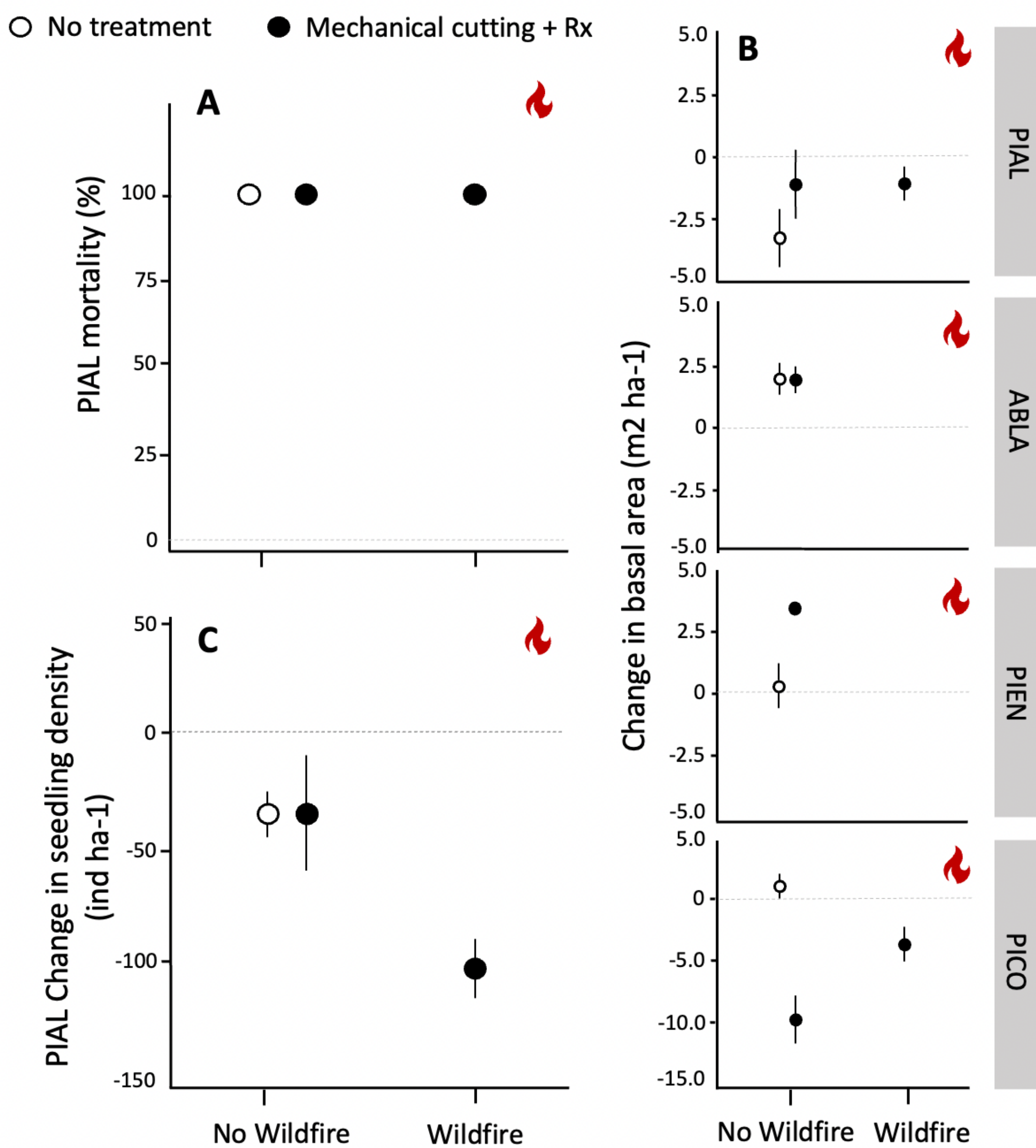

Figure 3. Effect of wildfire (x-axis; flame icon) after mechanical cutting and prescribed burning (solid circles, treated; open circles, not treated) on (A) mean (± 1 SE) mature *Pinus albicaulis* tree mortality (% over the 15 year period); (B) mean (± 1 SE) change (change pre to 15-years post treatment) in live tree basal area ($m^2 ha^{-1}$) of *Pinus albicaulis* and three shade-tolerant conifer species; and (C) mean (± 1 SE) change (change pre to 15-years post treatment) in *Pinus albicaulis* seedling density (ind ha^{-1}). PIAL = *Pinus albicaulis*, ABLA = *Abies lasiocarpa*, PIEN = *Picea engelmanii*, and PICO = *Pinus contorta*. Data are from Beaver Ridge.

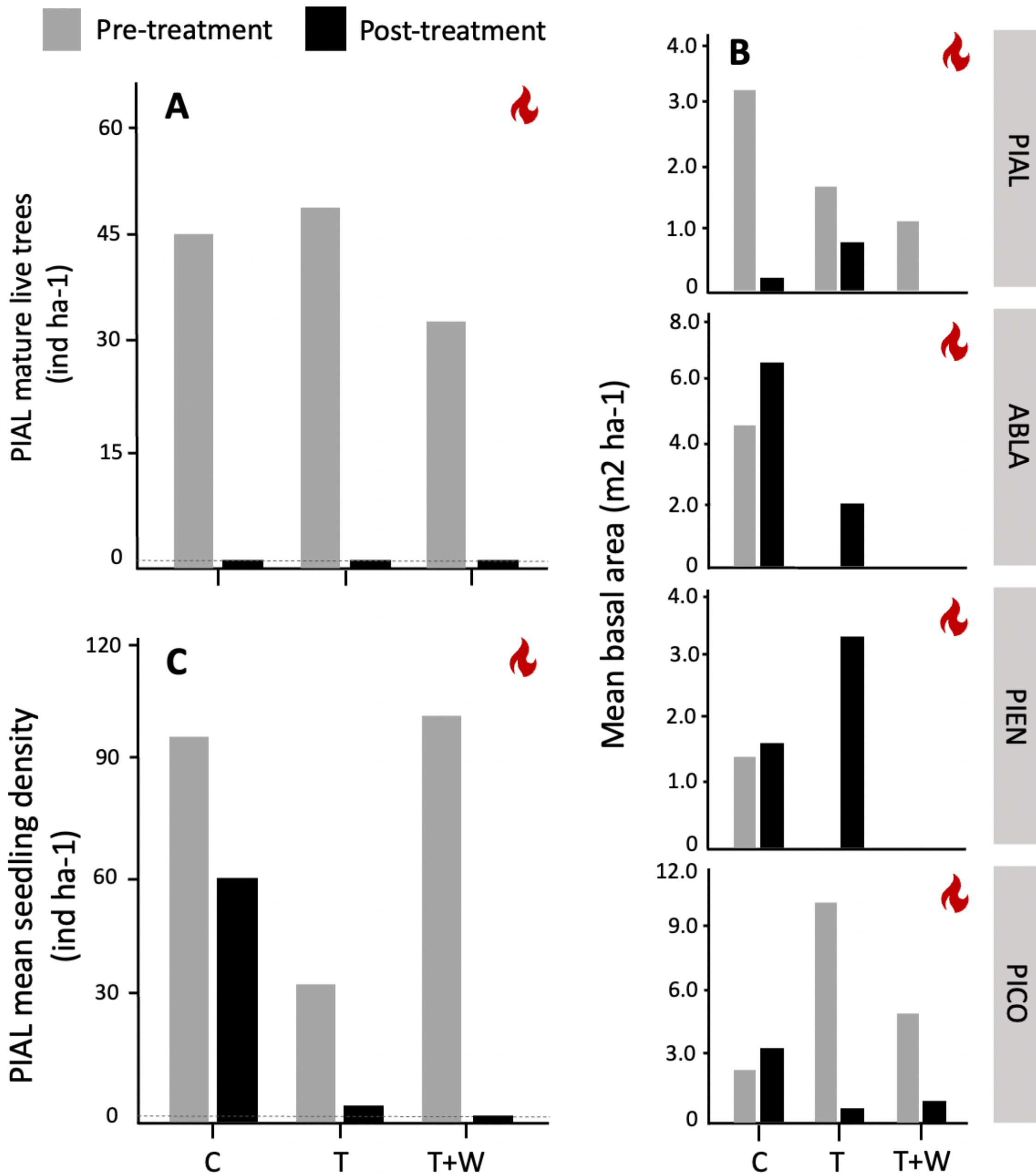

Figure 4. Effect of wildfire after mechanical cutting and prescribed burning, and treatment effect (gray, pre-treatment; black, post-treatment) on (A) density of live mature *Pinus albicaulis* tree (ind ha^{-1}); (B) mean basal area change ($m^2 ha^{-1}$) of *Pinus albicaulis* and three shade-tolerant conifer species; and (C) mean *Pinus albicaulis* seedling density (ind ha^{-1}). PIAL = *Pinus albicaulis*, ABLA = *Abies lasiocarpa*, PIEN = *Picea engelmanii*, and PICO = *Pinus contorta*. C = control, T= treated with prescribed burning and mechanical cutting, T+W= treated with prescribed burning and mechanical cutting and affected by wildfire. Data are from Beaver Ridge. Flame icon indicates units that experienced wildfire after treatment.

Over the 15-year period, density of *P. albicaulis* seedlings decreased in all stands: from 107 ind ha^{-1} (±22 SE) to 0 ind ha^{-1} for the prescribed burning and mechanical cutting stand affected by wildfire; from 39 ind ha^{-1} (±11 SE) to 4 ind ha^{-1} (±4 SE) for the prescribed burning and mechanical cutting stand not affected by wildfire; and from 94 ind ha^{-1}(±36 SE) to 60 ind ha^{-1} (±16 SE) for the untreated control stand. There was a statistically significant difference among units (H= 7.82, df= 2, *p*= 0.02) (figure 3C, figure 3C): the treated stands with and without wildfire varied significantly (*p*= 0.01), and there was also a significant difference between the treated stand affected by wildfire and the control (*p*= 0.01); however, change in *P. albicaulis* seedling density did not vary significantly between the treated stand not affected by wildfire and the untreated control stand (which also did not experience wildfire) (*p*= 0.91).

3.3. Individual tree mortality

The model of predictors of individual tree mortality resulted in a McFadden's Pseudo-R^2 of 0.17. None of the individual tree characteristics (DBH, tree height, LCBH, crown scorch) were significant predictors of *P. albicaulis* tree mortality. Although there was a trend towards a negative relationship between DBH and mortality (figure 7A), this relationship was not significant (*p*= 0.314). There was a significant effect of one of the three covariates on individual mature tree mortality (Table 3): relative basal area removed was significantly negatively related to mortality (*p*= 0.003) (Figure 7B). Each 10% increment of basal area removed pre-treatment is associated with a 14% decrease in odds of *P. albicaulis* mature tree mortality. The other two co-variates, stand condition (pre-treatment basal area) and area burned (%), were not significant predictors of mortality. Finally, there was a significant interaction between basal area removed and DBH (p= 0.003), but the interaction between DBH and area burned was not significant.

Table 3. Summary statistics for the effect of individual tree characteristics, stand conditions (pre-treatment basal area), and treatment intensity on individual tree mortality. β= coefficient estimates; SE= standard error; z= z-score; p= p-values (<0.05 bolded; insert interval italicized), BAR= Basal area removed (%); DBH= diameter at breast height; (cm) LCBH= live crown base height (m); PT BA= pre-treatment basal area ($m^2 ha^{-1}$).

Coefficients	β	SE	z	p
Intercept	3.401	1.58	2.14	**0.031**
DBH	-0.200	0.19	-1.07	0.314
Tree height	-0.032	0.04	-0.78	0.432
LCBH	-0.040	0.05	-0.76	0.441
Crown scorch	-2.400	1.94	-1.23	0.217
PT BA	-0.069	0.07	-0.89	0.369
BAR	-1.160	0.05	-2.87	**0.003**
Area burned	6. 492	5.81	1.11	0.264
BAR:DBH	0.014	0.01	2.91	**0.003**
Area burned:DBH	-0.554	0.56	-0.98	0.323

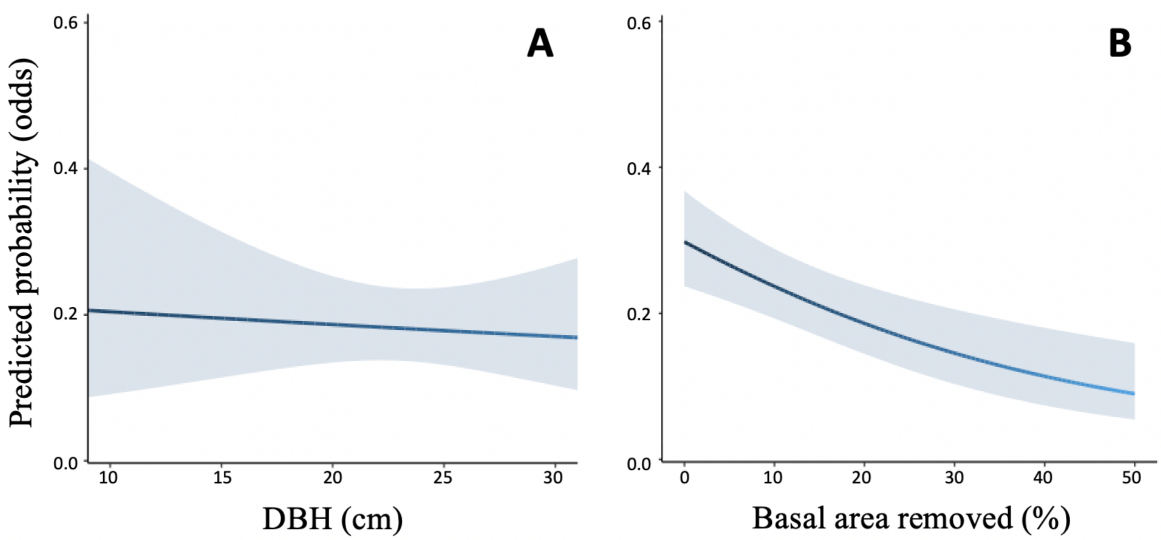

Figure 5. Predicted probability (odds) for *Pinus albicaulis* individual tree mortality for (A) diameter at breast height (DBH; cm), and (B) basal area removed (%). Confident intervals calculated with 68% confidence.

3.4.Efficacy of monitoring design

The precision of estimation (relative margin of error) achieved varied across response variables (figure 5A). The most precisely measured variable was basal area ($m^2 \cdot ha^{-1}$) of *P. albicaulis*, which was estimated with a relative margin of error of 34% (±4 SE). Density of *P. albicaulis* seedlings and % *P. albicaulis* tree mortality were estimated with even more error (relative margin of error = 71% (±7 SE) and 47% (±15 SE), respectively). To achieve a 20% relative margin of error would have required sampling 30, 130, and 65 plots (compared to the 10 plots that were measured per stand) to capture *P. albicaulis* basal area, seedling density, and mature tree mortality, respectively (figure 5B).

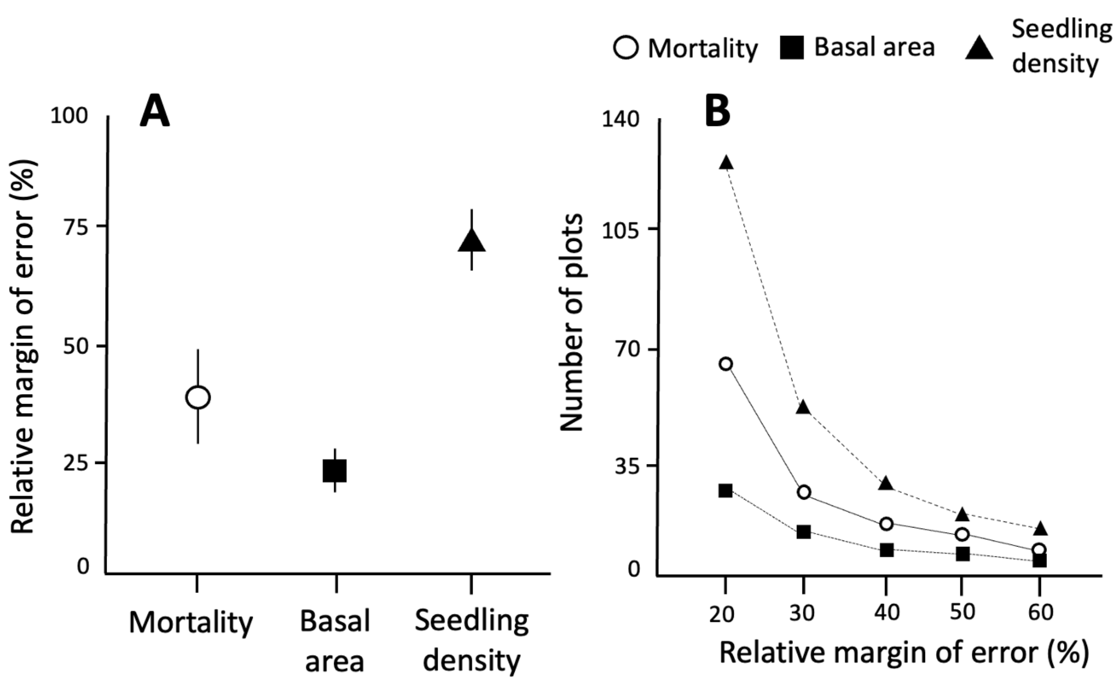

Figure 6. (A) Pre-treatment relative margin of error (%) achieved for *Pinus albicaulis* response variables (plot-level tree mortality (%), live tree basal area ($m^2 ha^{-1}$), and seedling density ($ind\ ha^{-1}$). (B) Relationship between sample size (number of plots within sites) and relative margin of error around the mean for *Pinus albicaulis* response variables.

4. Discussion

Over the past several decades, there has been an increase in the scope and magnitude of ecological restoration projects across multiple ecosystems (Bernhardt et al. 2005; Schoennagel et al. 2009; Nunez-Mir et al. 2015), and a corresponding increase in interest in assessing treatment efficacy and effects (Osenberg et al. 2006; Larson et al. 2013; Nelson 2021). For forested ecosystems, restoration practitioners and land managers need information on how changing environmental conditions and stochastic events, such as wildfire, can impact ecological responses to treatment, as well as the capacity of broadly implemented study designs to detect responses. My findings, however, reveal the difficulty in assessing forest responses to restoration treatments over time. Although the RWPE project utilized best practice design for testing treatment effects (Osenberg et al. 2006; Nelson 2021) and had a relatively large number of study sites (n=5), my ability to make inference was limited due to loss of experimental units to wildfire, the confounding effects of *D. ponderosae* and *C. ribicola*, and low precision of estimation, especially in measurements of seedling density. The only detectable responses to treatment were changes in basal area of *P. engelmannii* and *P. contorta,* although responses varied by treatment type, and that mechanical treatment may reduce mature *P. albicaulis* tree mortality. Additionally, I found wildfire-related declines in *P. albicaulis* seedling density. The low precision in my estimates of all variables of interest (mortality, basal area, and seedling density) coupled with lack of statistical power indicates the need for higher replication of both experimental sites and subsamples within experimental units when monitoring variables with high spatial and temporal heterogeneity.

4.1. *P. albicaulis* mortality and abundance

Given that *P. albicaulis* is a threatened species, understanding the effect of management treatments on *P. albicaulis* trees is critical to making informed management decisions. There is growing concern about potential negative or unexpected effects of treatment for restoration purposes including potential for increased tree mortality (Maher et al. 2018; Nelson & Keville 2018), especially after prescribed fire (Hood et al. 2008; Nelson & Keville 2018; Cansler et al. 2020). In fact, Keane and Parsons (2010) documented for my study sites that mature tree mortality within the first 5 years post-treatment was predominantly caused by damage from prescribed burning. However, this trend was no longer evident by the 15th year of measurement: I did not detect increased *P. albicaulis* mature tree mortality in treated compared to untreated stands, and my model of predictors of individual tree mortality failed to detect a significant effect of prescribed burning. Nonetheless, inference from this study is limited given that both sites used to assess mortality, Bear Overlook and Beaver Ridge, had been affected by *C. ribicola* infection and a regional *D. ponderosae* outbreak (Bentz et al. 2011), limiting my capacity to separate the effect of treatment from *D. ponderosae* and *C. ribicola* induced mortality.

Although I did not find significant trends with respect to prescribed fire, I did find evidence that thinning for competitive release may increase the odds of individual tree survival, consistent with results of Hood and collaborators' (2016) modelling study. Specifically, I found a significant interaction between basal area removed and DBH (p= 0.003), indicating that larger trees were less susceptible to mortality with higher level of basal area removal, potentially due to growth release that occurred after thinning competing conifers. Although I did not specifically measure growth release, previous studies have found clear evidence linking thinning to increased growth: Retzlaff and collaborators (2018), working on the same study sites as those that I assessed, found that growth rates for live *P. albicaulis* saplings was about 3-times higher in stands with mechanical cutting and prescribed burning than in control stands 5 years after harvest; in addition, Keane and collaborators (2007) found that on sites across Montana thinning led to higher rates of growth for remnant mature *P. albicaulis* trees. However, growth release after thinning has not been consistently observed: a study on the ecological effects of silvicultural treatments on *P. albicaulis* at five sites across the western US found evidence of growth release and reduced rates of mortality of *P. albicaulis* at only one site. Furthermore, the authors attributed the response at this site to reduced beetle-induced mortality due to reduction in density of host trees (*P. contorta*) (Maher et al 2018).

Pinus albicaulis is experiencing high rates of mortality across the western Unites states due to both *D. ponderosae* outbreaks and *C. ribicola* infection. The sites included in this study were affected by a severe region-wide *D. ponderosae* outbreak during the study period (Bentz et al. 2011), however there was no available information about beetle impacts on the study sites. In addition, high levels of *C. ribicola* infection were documented at the site level before treatment implementation: pre-treatment infection rates were around 70% and 51% for Bear Overlook (site not affected by wildfire) and Beaver Ridge (site affected by wildfire), respectively (Keane & Parsons 2010). Although information on site-level infection rates were collected pre-treatment, data on *individual* tree infection were not and, therefore, I was not able to attribute individual tree death post-treatment to *D. ponderosae* or *C. ribicola*.

Schoettle and Sniezko (2007) discussed the need for proactive management (such as the restoration treatments implemented in this study) to mitigate the ecological effects of *C. ribicola* and prevent its invasion, including: (1) managing forest composition, (2) increasing tree vigor, and (3) diversifying age class structure – all of which form part of the treatment objectives implemented at the site (Keane & Parsons 2010). However, the observed increase in *P. albicaulis* mature tree mortality across sites suggests that a stand-health threshold had been crossed, which may have limited the ability of recommended treatments to improve ecological integrity (Schoettle & Sniezko 2007; Tomback et al. 2022). Thus, the severity of rust infection and stand health should be considered in planning *P. albicaulis* restoration treatments.

4.2 Abundance of competing conifers

One of the primary reasons for doing restoration treatments in *P. albicaulis* forests is to release remaining *P. albicaulis* trees from competition with shade-tolerant conifers (Keane et al. 2012; Tomback et al. 2022). However, there have been few studies that have effectively assessed the impact of treatments on competitor species (Keane & Parsons 2010; Keane et al. 2012; Maher et al. 2018). Of the three competing conifer species included in this study, I was able to detect a response to treatment for two, *P. engelmannii* and *P. contorta,* although for both effects varied significantly by treatment type. *Picea engelmannii* increased across all treatment units at Bear Overlook, but the increase was greater in the burning-only unit relative to the unit that had the combined burning and mechanical cutting treatment, and neither increases were different from background levels (i.e. the increases observed in the control). The higher increase in basal area in the burn only unit is likely because the burning in this unit covered less area compared to the more widespread burning that occurred in the prescribed burning and mechanical cutting unit. Because of this, more individuals survived the treatment in the burn-only unit and, therefore, increased in basal area. In addition, some seedlings from the pre-treatment sampling grew enough over time to be counted as saplings in the post-treatment measurement.

Pinus contorta basal area response also differed by treatment type. At Bear Overlook (the site without wildfire), basal area decreased in the prescribed burning only unit, where fire killed a large number of small trees and where there was limited post-treatment regeneration. Conversely, at the same site, basal area increased in the prescribed burning and mechanical cutting unit, even though the area burned was greater in this unit than it was in the burn-only unit. This trend may be due to the fact that in the unit with the combined treatment there wasabundant post-treatment regeneration of *P. contorta* experienced, as well as a large number of seedlings that survived treatment and grew enough to be counted as saplings. Similar increases in *P. contorta* basal area have been described after mechanical-cutting-only treatments conducted for fire-mitigation and restoration in *P. albicaulis* forests (Maher et al. 2018).

Reduction of *A. lasiocarpa* has been noted as an important restoration objective (Keane et al. 2012; Tomback et al. 2022), given concern about shifts in composition towards *A. lasiocarpa* in the absence of disturbance (Arno 2001; Keane 2002), although this trend has not always been found (Amberson et al. 2018). However, Maher et al (2018) documented that thinning may actually increase regeneration of *A. lasiocarpa* in *P. albicaulis* stands. The fact that I observed increased basal area of *A. lasiocarpa* across all units, regardless of treatment, adds evidence that the use of silvicultural treatments combined with prescribed burning may not be effective at releasing *P. albicaulis* stands from encroaching *A. lasiocarpa*.

Regarding treatment response to wildfire, I was only able to assess the effect for one competing conifer, *P. contorta,* since *A. lasiocarpa* and *P. engelmannii* were not present in the treated stands affected by wildfire. At Beaver Ridge, both treated units, with and without wildfire, experienced greater decreases in basal area of *P. contorta* than did the untreated control that did not burn by wildfire. The declines in *P. contorta* basal area were not different between the treated stands affected and not affected by wildfire. This lack of difference between these units was largely driven by high rates of mortality of *P. contorta* within the prescribed fire stand that did not burn by wildfire; by random chance, most *P. contorta* trees were located within plots affected by the prescribed fire. On the other hand, I observed increases of *P. contorta* basal area in the untreated control stand, as the trees were able to put on 15 years of growth. In addition, basal area may have increased more than expected due to a potential resource release from the observed high *P. albicaulis* mortality.

4.3 Regeneration

Across study sites, there was a large decline in seedling densities over the 15-year period, even at the control sites. This trend may be due to the high rates of rust infection and beetle attacks across the sites (McKinney & Tomback 2007; Leirfallom et al. 2015; Shepherd et al. 2018), which may be reducing cone and seed production, and disrupting Clark's nutcracker caching patterns in the region (Larson & Kipfmueller 2010). Although lack of nutcracker caching could be a limiting factor in regeneration, the field crew observed frequent nutcracker caching during sampling across years (Keane, *pers. com.*). Cone and seed production may also be adversely affected by changing climatic conditions, which may affect frequency of masting events (Larson & Kipfmueller 2010; Crone et al. 2011).

Given increasing frequency and severity of wildfire events, it is important to understand the effects of wildfire on regeneration and whether pre-wildfire treatment alters that effect. Despite high margin of error, I did find significantly greater declines of *P. albicaulis* recruitment in the Beaver Ridge stands that had been treated and affected by post-treatment wildfire compared to stands (treated and untreated) not affected by wildfire.

Although I found an effect of wildfire at Beaver Ridge, I was not able to detect differences in seedling density in response to either of the mechanical cutting treatments, *slashing* (Bear Overlook) and *nutcracker openings* (Beaver Ridge), despite the fact that *nutcracker openings* were designed to create suitable caching conditions (Keane & Parsons 2010). There are at least two possible explanations for lack of observed differences: first, even though the treatment created caching habitat, it may not have addressed other limiting factors in seedling recruitment, such as availability of nutcrackers or suitable microsite conditions (such as canopy protection and neighboring vegetation and structure) for seedling establishment (Perkins 2015; Amberson et al. 2018). On the other hand, lack of observed response could be due to the fact that the sampling design was not sufficient to precisely estimate seedling densities. Seedling density estimates had relative margin of error of ca 70%, highlighting the challenge of monitoring natural regeneration. Although for some study variables it is possible to detect trends by measuring change over time at the plot level, detecting trends in seedling density requires suitable estimates of stand-level means given rapid turn-over of seedlings. Furthermore, estimation of seedling density is particularly difficult for species like *P. albicaulis*, whose regeneration dynamics are highly variable not just in time but also in space (Lorenz et al. 2011; Barringer et al. 2011; Leirfallom, et al. 2015).

4.4 Study design limitations

Even though the RWPE project was the first and most comprehensive study of restoration treatments in *P. albicaulis* forest, the ability to assess effectiveness of the treatments was limited by the large number of sites that were disturbed by wildfire, beetles, and rust, coupled with lack of consistent data collection from control and treated sites within the same sample year, and too limited a number of subsamples to adequately estimate key study variables.

The fact that a study of responses to forest management treatments with five sites, and multiple treatment units within sites, did not have enough replication is sobering, given the difficulty inherent in establishing study sites for a BACI design. For this study, the number of possible sites was limited, given the need for treatment units to be on Forest Service land and close to roads or trails for crew transportation. In addition, all treatments had to go through the federal approval process and comply with individual ranger district requirements. Even after approval, the implementation of treatments was contingent on suitable weather conditions during the treatment implementation window (Keane & Parsons 2010; Keane & Parsons 2010). These limitations reduced the number of sites available to include in the study. In addition, the loss of multiple sites to unplanned wildfires reduced even more the sites available to test response to treatment. Low site replication has been a common issue even for large-scale experimental studies: for instance, 11 of the 13 areas included in the Fire and Fire Surrogate Study (FFS) were implemented at a single site with different levels of replication of treatment units (Schwilk et al. 2009).

Another issue beyond the total number of sites is the lack of consistent data collection from control and treated sites within the same sample year. Even though the original project included five sites, only two of the five original sites had data collected from both treated areas and controls by the 15th year after treatment implementation, reducing the capacity to assess the long-term effect of treatment and treatment response to unplanned wildfire. Having had that data from all sites and each monitoring year would have increase the level of replication and also would have allowed a more rigorous survival analysis by attributing mortality to specific post-treatment years.

The fact that I found low precision of estimation for measurements of study variables highlights the importance of including a sufficient number of subsamples within each treatment unit when assessing treatment effects. Although 10 plots per stand did not produce reliable estimates of mean stand conditions for my study variables, this number of subsamples is on the high end relative to other studies of the effects of forest management treatments. For instance, a review of forest monitoring practices (Foster 2001) found that most studies on the ecological effect of management treatments have fewer than 10 subsamples per experimental units. Similarly, the use of 10 subsamples or less is the standard in some high-profile experiments on the effects of thinning and burning (McIver et al. 2012; Stephens et al. 2012; Briggs et al. 2017) such as the Fire and Fire Surrogate Study (Schwilk et al. 2009) and the Colorado's Front Range Collaborative Forest Landscape Restoration Program (CFLRP) (Briggs et al. 2017). These studies also were not able to detect significant differences among treated and controls units for some study variables. On the other hand, there is evidence that use of additional subsamples can increase power of detection; for instance, the DEMO Study (Halpern et al. 1999) designed to test different levels and patterns of green-tree retention used 17 plots per experimental unit and resulted in ability to make inference across most of their study variables.

5. Conclusion and management implications

My findings confirm the complexity of designing effective monitoring studies to assess ecological responses to widely used restoration treatments, and that even long-term, well-replicated BACI designs may fail to detect an effect. Although I did not find significant effects of treatment or post-treatment wildfire on *P. albicaulis* mature tree mortality, basal area change, or seedling density change 15 years post-treatment, it is unclear whether my results are driven by lack of effects or due to limitations of the study design. Additionally, both sites were affected by *D. ponderosae* damage and *C. ribicola* infection, which also may have limited ability to detect a treatment response. Thus, it is crucial for continuing ecological research and management on *P. albicaulis* forests, as well as other forest types, to integrate inherent ecosystem complexity into monitoring efforts so that they don't fall short. Finally, given that data needs for inference may be too large for any one administrative unit or research project alone, consideration should be given to developing a large-scale long-term monitoring network and designate core areas for restoration to improve understanding of the efficacy and effects of restoration treatments in *P. albicaulis* ecosystems.

The most important lessons learned from this long-term monitoring project are:

- Implement and monitor treatments using appropriate experimental design and adequate replication to allow inference. If there is high risk of loss of experimental stands or sites due to disturbance, extra sites may be needed to allow for adequate replication over time. Working within the context of a monitoring network (i.e. teaming up with other projects) may facilitate achieving required levels of replication.
- Always measure control units every year in which treatment units are sampled. This will allow managers to separate the effects of treatment from site-to-site variation.
- If the purpose of monitoring is to determine effects of treatments on natural regeneration, use a sufficiently high number of subsamples and bigger plot sizes to allow for precise estimates of mean stand seedling density.
- Prior to including sites in studies of treatment effects, assess risk of rust infection and beetle attack, in order to avoid confounding treatment response with impacts of

insects and disease. Stand health also may be a consideration for prioritizing where to implement proactive restoration treatments; it might be better to prioritize treatments in stands with low or no damage by *C. ribicola* and/or *D. ponderosae*.

- The fact that single treatments were not effective at achieving long-term restoration objectives (such as reducing competing conifers), suggests that multiple treatments may be needed.

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