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Effects of post-fire management on dead woody fuel dynamics and stand structure in a severely burned mixed-conifer forest, in northeastern Washington State, USA

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A B S T R A C T

The increasing amount of high-severity wildfire in historical low and mixed-severity fire regimes in western US forests has created a need to better understand the ecological effects of different post fire management approaches. For three different salvage prescriptions, we quantified change in stand structural metrics (snag densities and snag basal areas), dead woody fuel loadings, tree regeneration survival, and percentage change in vegetation cover before and after post-fire logging 1 year after the 2015 Stickpin Wildfire on the Colville National Forest in northeastern Washington State, USA. In a generalized randomized block design three salvage logging prescriptions were randomly assigned within each block: no treatment control (C); standard salvage retention (SSR; thin to 3.4 m^2/ha basal area); and mimic green tree thinning (GTR; thin to 10.3 m^2/ha basal area). SSR reduced average snag basal area 73–83% to 4.1–8.8 m^2/ha (68–674 trees ha^{-1}). GTR reduced average snag (standing dead trees) basal area 41–71% to 6.5–15.9 m^2/ha (90–794 trees ha^{-1}). There were mixed results for the change in dead woody fuel loadings depending on fuel size class. In general, fine (FWD) and coarse woody (CWD) debris tended to increase immediately post-treatment in logged areas relative to the controls but did not exceed management loading threshold for providing acceptable risk of fire hazard. Treated stands had a significant increase in FWD relative to controls, including the individual 1-, 10-, and 100-hr fuel size classes. The treatment effect differed by experimental block. The 1000-hr sound class did not have a significant treatment effect. Changes in surface fuel loading were inconsequential to modeled wildfire behavior metrics (rate-of-spread, flame lengths). The Fire and Fuels Extension to Forest Vegetation Simulator (FFE-FVS) modeling projected CWD accumulation in the controls exceeded total accumulation in both treatments. Future fuel loadings may affect reburn severity as our simulated wildfire 20 years after harvesting caused significant mortality (89%) to regenerating forest. Almost all blocks showed a decrease in seedling counts pre and post-logging, including the control plots. This study provides empirical data on the effects of different post-fire management strategies that can inform environmental analyses for future post-fire management decision and address social concerns associated with this often-controversial practice (Roccaforte et al., 2012).

1. Introduction

Over the past few decades, the frequency and size of contemporary wildfires has dominated the national news headlines and are at the center of a political debate on the management of our national forests following stand-replacing wildfires (Beschta et al. 1995, Lindenmayer et al. 2008, Peterson et al. 2009). With projected warming climate, wildfires are predicted to increase in frequency, extent and severity in the western United States (Westerling et al. 2006; Littell et al. 2009; Dennison et al. 2014), particularly in flammability-limited systems (Littell et al. 2018). The increase in fire severity and frequency are due to a century of fire exclusion, historic timber harvest that removed large fire-resistant trees, livestock grazing, and fire suppression (Hessburg et al. 2005). This has changed the forest structure, disturbance regime, and specie composition of many dry-coniferous forests (Agee and Skinner 2005). Compared to historical conditions, modern dry forests (dry ponderosa pine and mixed-conifer forests) have higher stem densities (Hessburg et al. 2005; Harrod et al. 2009; Hagmann et al. 2014) and increased woody fuel loadings (Agee and Skinner 2005). These forests are susceptible to uncharacteristic, stand-replacing wildfires, which produce large mosaics of fire-killed trees (Stephens et al. 2009; Stevens et al. 2017). These snags (standing dead trees) are an important and integral component of a healthy, functioning forest ecosystem (DellaSala et al. 1995). Snags provide wildlife habitat (Castro et al.

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2010; Ritchie et al. 2013) and are essential for post-disturbance forest recovery and biodiversity (Lindenmayer et al. 2008). However, fire and forest managers have prominent management concerns about the fuel trajectories of dead trees under changing climate (Littell et al. 2009; Coppoletta et al. 2016), particularly fuel accumulation following larger wildfires setting up conditions for future high severity fires (Thompson et al. 2007). Post disturbance management of large areas with fire-killed trees is a central management concern for low severity, high fire-frequency forest regimes (Coppoletta et al. 2016).

Following large wildfires on federal, state, and private forest lands, the conventional policy is to initiate post-fire management activities, which includes the harvest of a portion of fire killed trees to recuperate the economic value of merchantable trees (USDA 1996; McIver and Starr 2001; Nemens et al. 2019). This management practice, known as “salvage logging” is a contentious issue on federal lands (Beschta et al. 1995, McIver and Starr 2001) and is extensively implemented worldwide (Castro et al. 2010, Thorn et al., 2018, Leverkus et al. 2018). Salvage logging (ground-base) requires the use of heavy mechanical equipment to extract timber (remove snags); as such, salvage logging critics believe harvesting fire-killed trees impedes natural forest recovery (Lindenmayer et al. 2008), reduces critical wildlife habitat for species such as the black-back wood pecker (*Picoides arcticus*; DellaSala et al. 1995), increases soil erosion (Karr et al. 2004), increases invasive species (Beschta et al. 1995), reduces species richness (Sexton 1998), increases dead woody fuel loading, subsequent fire hazard, and reduces forest regeneration (Martínez-Sánchez et al. 1999, Donato et al. 2006, Leverkus et al. 2012). Critics hold that salvage logging is, for the most part, economical and not ecological (Beschta et al. 1995). Conversely, advocates for the practice argue in addition to recuperating the economic value of fire-killed timber resources (Barker 1989), salvage logging reduces fuel accumulation (Neeman et al. 1997, Peterson et al. 2015), improves public and firefighter safety (Passovoy and Fulé 2006) facilitates fire suppression (Jenkins et al. 2012), reduces insect epidemics (Amman and Ryan 1991), accelerates forest reforestation with desired tree species stocking (Sessions et al. 2004; Collins and Roller 2013), and mitigates negative consequences of reburns (Thompson et al. 2007; Coppoletta et al. 2016). Dead woody fuel succession is a positive feedback mechanism for subsequent wildfires, or reburns (Brown et al. 2003). The “reburn hypothesis” is the assertion that large areas with fire-killed trees are prone to future high-severity wildfire as dead woody fuels accumulate and shrubs regenerate over time (McIver and Starr 2001).

Replicated longitudinal experiments investigating the effects of post-wildfire salvage logging are scarce (McIver and Starr 2001; Ritchie et al. 2013). Few empirical studies have monitored fire-killed areas over time with the objective of quantifying post-fire fuel dynamics and the potential effects of a reburn (McIver and Starr 2001). Most scientific data on salvage logging effects and long-term fuel succession dynamics is derived from observational and unreplicated studies such as chronosequence sampling (Passovoy and Fulé 2006; Peterson et al. 2015) or simulation model projections (Dunn and Bailey 2012), both of which have limitations and assumptions (Brown et al. 2003; McIver and Ottmar 2007; McGinnis et al. 2010). Chronosequence sampling assume space-for-time substitution to infer temporal dynamics (Johnson and Miyanishi 2008) and simulation models are limited by modeling assumptions used to make predictions (Cruz and Alexander 2010).

Contemporary large stand-replacing wildfires provide ample opportunities to understand short-and-long-term effects of salvage logging and fuel succession. However, funding, administrative logistics and coordination with a collaborating agency make establishing such studies a major challenge (Ritchie and Knapp 2014). In 2016, we had the rare opportunity to collaborate with natural resource forest practitioners on the Colville National Forest in northeastern Washington State and the Northwest Washington Forestry Coalition to design and implement a project to quantify the effects of different post-fire management approaches. The site was part of the Colville National Forest Deer Jasper hazardous fuel and forest restoration project (USDA 2014) that had been laid out and was ready for implementation just before the 2015 Stickpin wildfire burned the site as high severity (USDA 2016). The vision of the Second Creek Post Fire Recovery Categorical Exclusion project was to create a demonstration site for field trips and shared learning to inform the post-wildfire management debate (USDA 2016). In particular, stakeholders were interested in understanding salvage logging effects on woody fuel loadings, seedlings, shrubs and postfire fuel dynamics following different basal area retention (snag densities) strategies.

In this study, we quantified the effects of salvage logging on snag densities and biomass, dead woody fuel loadings, tree seedling survival, and non-tree vegetation cover in dry coniferous forests in northeastern Washington State. Our study objectives were to: (1) use pre and post-empirical field measurements to quantify short-term (1-year post-harvest) effects of salvage logging on stand structure, fine woody debris, and coarse woody debris, percentage shrub cover, tree seedling/regeneration survival as influenced by variable density thinning prescriptions; and (2) use the Fire and Fuel Extension to the Forest Vegetation Simulator (hereafter FFE-FVS; Rebain 2010, Crookston and Dixon 2005) to predict long-term fuel succession dynamics and future fire behavior (reburn hypothesis). The initial intent of the variable-density thinning prescriptions was to produce post-harvest experimental units with different residual basal area retentions to provide opportunities to quantify short-and-long-term dead woody fuel dynamics (succession) and to quantify snag decomposition. We tested three hypotheses: (1) post-fire logging would initially increase dead woody fuels (FWD and CWD) relative to controls (unlogged stands) by transferring non-merchantable woody debris from tree canopy to the forest floor; (2) post-fire logging reduces long-term fuel loadings and potential fire severity by removing dead trees (snags); (3) post-fire logging operations would reduce tree regeneration and percentage of non-tree vegetation.

2. Material and methods

2.1. Study area

The study was located on the 445,154 ha Colville National forest in northeastern Washington State, USA (latitudes between 48.8832245 and 48–53′00″N and longitudes between 118.7678127 and 118–46′04″W (Fig. 1). The forest lies with the Okanagan Highland geologic province characterized by moderate slopes with rounded summits (USDA 2014). Climate has both maritime and continental characteristics because air masses from the continent and proximity to the Pacific Ocean (Phillips and Durkee 1972). The western zone of the Colville National forest is in a rain shadow formed by the North Cascades Mountains, while the eastern zone has a moist near-maritime climate (Williams et al. 1995). The majority (80%) of annual precipitation falls as snow from September to May at higher elevation 1219–1524 m. Annual precipitation is highest in winter and spring and ranges from 38 to 50 cm in valleys and 76–102 cm in mountains. In summer months (July to September), valley temperatures range from 4.4 to 27 °C. Winter temperatures range from −12 to −6 °C (Phillips and Durkee 1972). Topography is characterized by north–south trending 1524–2133 m elevation mountain ranges with the intervening valleys of the Pen Oreille, Columbia, Colville, Kettle and San Pol rivers (Williams et al. 1995). The dominant soil orders are inceptisols and andisols followed by mollisols and alfisols (Farr et al., 2017). Soils consist of moderately-well to well-drained soils formed in the volcanic ash overlying glacial till or granitic, gneissisc, andesitic, rhyolitic, or schist bedrock formations in the higher elevation forested areas. Soils were formed from glacial till, outwash, lacustrine deposits, colluvium and alluvium (Williams et al. 1995). Vegetation type characterized as warm-dry Douglas-fir shrub and cool mesic Douglas-fir, grand fir forb-shrub (Williams et al. 1995). The understory varies from graminoid to

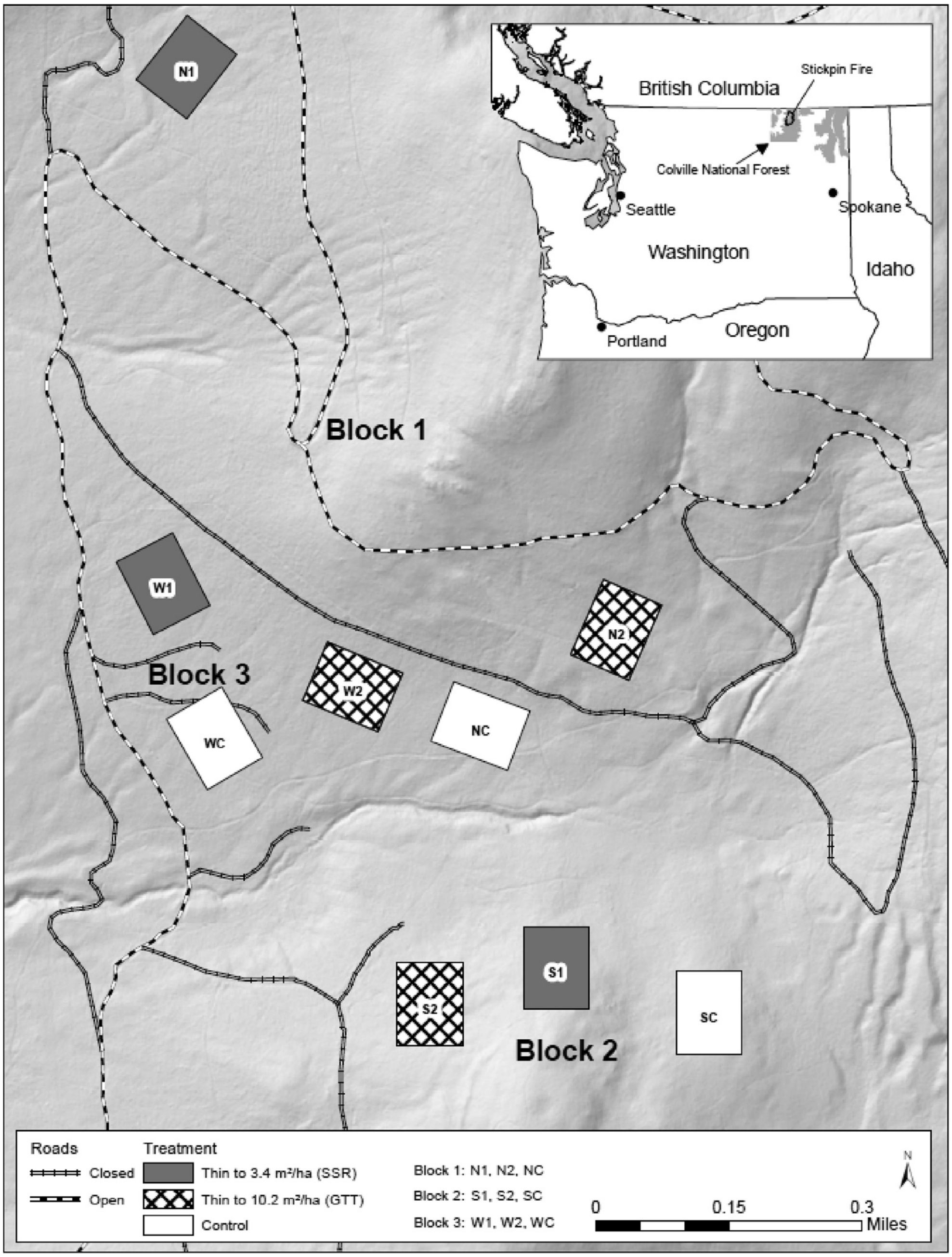

Fig. 1. Location of the 9 experimental units within 3 Blocks with the Stickpin Wildfire, Colville National Forest in northeastern Washington State, USA. For each block, three salvage logging prescriptions were assigned randomly to experimental units for a complete randomized-block design: 1) unlogged control (C); 2) thin to 3.4 m^2/ha basal area (SSR); and 3) thin to 10.3 m^2/ha basal area (GTR).

evergreen or deciduous shrubs and forbs. The forest type is predominantly Douglas-fir (Pseudotsuga menziesii (Mirb.) Franco) and western larch (*Larix occidentalis* Nutt.) with a lesser amount of lodgepole pine (Pinus contorta-Douglas ex Loudon) and ponderosa pine (*Pinus ponderosa* Lawson & C. Lawson) (Pass and Corvino 2014). Engelmann spruce (*Picea engelmannii* Parry ex Engelmann) and sub-alpine fir (*Abies lasiocarpa* (Hook.) Nutt.) are species common on north facing slopes and riparian draws. The fire regime condition class is high-frequency (0–35 years) low to mixed severity (Curtis and Wynecoop 2017; Pass and Corvino 2014).

2.2. Sampling design

The study sites were located within the 101 ha Second Creek Post Fire Recovery project area established in ~ 60-year-old, mixed-conifer plantations (average tree DBH 25.4–50.8 cm; canopy cover 60–80%). In 2015, the Stickpin Fire burned 21,901 ha from 11 August to 7 September 2015. The fire created large mosaics of high mortality (98–100% basal areas (BA) mortality) throughout the project area. Overall, Stickpin post-fire burn severity classes, based on percentage basal area loss as indicated on the Rapid Assessment of Vegetation Condition after Wildfire (RAVG) were: 46% high (75–100%), 18% moderate (25–75%), 19% low (< 25%), and 6% unburned (USDA 2016).

For this study we implemented a generalized randomized block design (GRBD) with replication. We randomly selected three blocks in the study area in areas classified as high burn severity (100% basal area mortality) with similar stand conditions and vegetation type. The

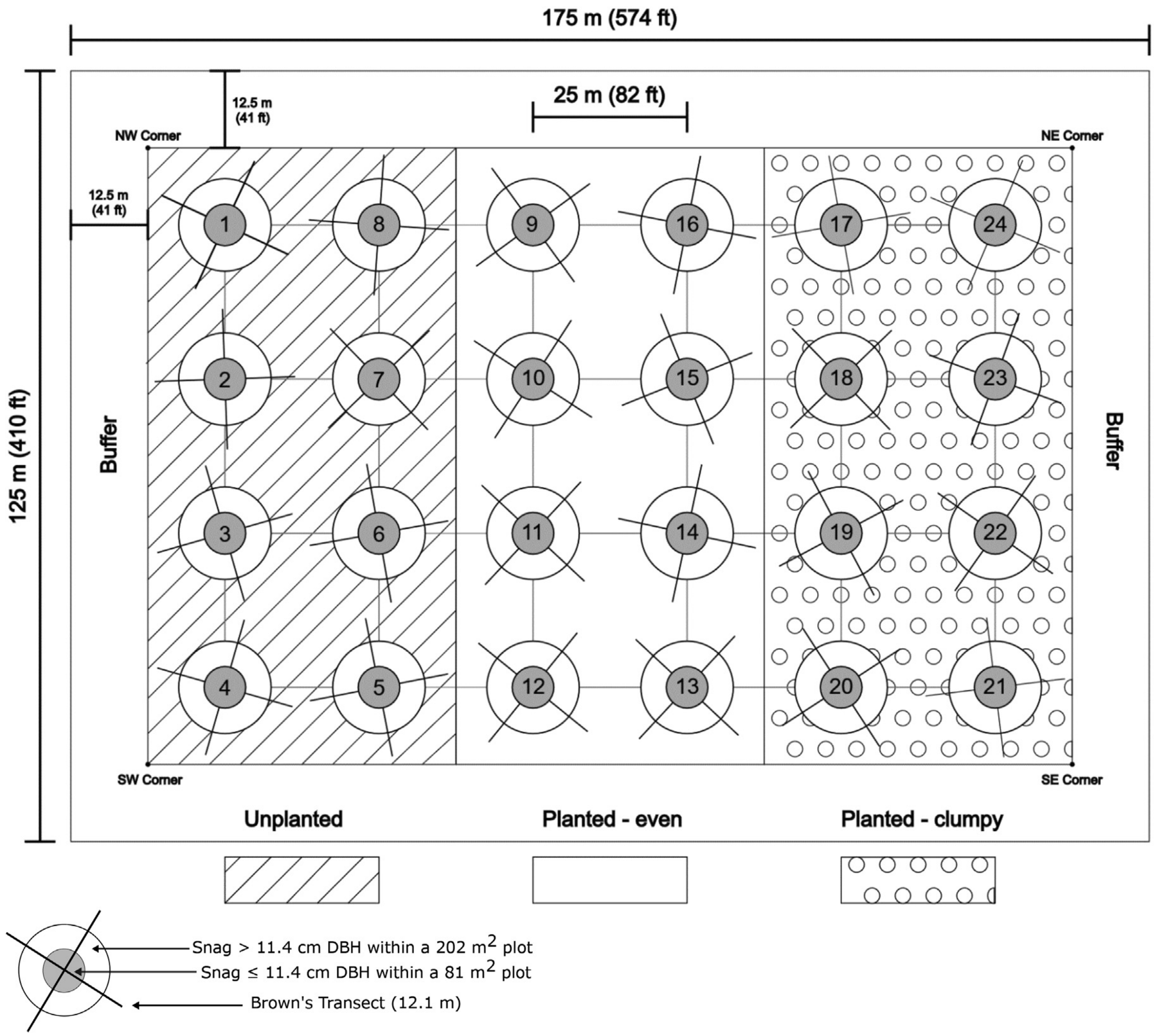

Fig. 2. In each Stickpin Wildfire experimental unit, a permanent monumented-grid of 24 nested fixed-area circular plots (216 total) were established to quantify pre and post-logging snag density (trees ha^{-1}), snag biomass (m^2/ha), fine (FWD < 7.62 cm diameter), and coarse (CWD > 7.62 cm diameter) dead woody fuel loadings (Mg ha^{-1}). Seedling density (trees ha^{-1}) and percentage non-tree vegetation coverage using 2.25 m^2 square quadrats installed every 5 m along 100 m transects (transects and quadrats not shown in figure).

general principle of GRBD is that stands within each block will be correlated with each other due to spatial proximity. The blocks themselves will be independent of each other due to random selection. Within each block we randomly assigned salvage logging treatments to 2.2 ha (125 × 175 m) experimental units, each with a 30 m buffer to avoid edge effects from roads or other treatment types. The treatments randomly assigned to each block were 1) No Treatment Control (C); 2) Standard Salvage Retention (SSR; thin to 3.4 m^2/ha basal area; and 3) Mimic Green Tree Retention (GTR; thin to 10.3 m^2/ha basal area). Whole-tree harvesting methods were used by which snags were felled with a mechanical feller-buncher and the entire snag was skidded to a central processing area where the branches were removed. No additional slash reduction treatments were implemented in the treatment areas. In spring 2018, logged experimental units were replanted to variable densities.

In each experimental unit, a permanent monumented grid of 24 nested fixed-area circular plots installed on a 25 m apart (216 total; Fig. 2) were established to quantify pre- (summer 2016) and post- (summer 2017) logging snag density (trees ha^{-1}), snag biomass (m^2/ha), fine (FWD < 7.62 cm diameter) and coarse (CWD > 7.62 cm diameter) dead woody fuel loadings (Mg ha^{-1}), seedling density (trees ha^{-1}), and percentage non-tree vegetation coverage. At plot center, we recorded species, diameter at breast height (DBH, cm), total height (m), and tree crown base height (CBH, m) for each snag > 11.4 cm DBH within a 202 m^2 circular, fixed-area plot. Data for snags ≤ 11.4 cm DBH were recorded on a 81 m^2 plot. FWD and CWD loadings were estimated using the planar-intercept method as described by Brown (1974) on four 12.1-m planar transects originating from random azimuths (Fig. 2). The first azimuth was random (random azimuth table built *a priori* in the office) and the other three were established by adding 90 degrees to the first azimuth. The end point of each woody transect was marked, first with a stake chaser whisker (temporary marker) attached to a 15.2 cm steel nail, and later (post-harvest) permanently marked with a 0.91 m aluminum conduit. Loadings were recorded by size class (i.e., time lag class: 1-h, 0–0.64 cm diameter; 10-h, 0.64–2.54 cm; 100-h, 2.54–7.62 cm; 1000-h, > 7.62 cm) corresponding to fuel moisture classes used for fire behavior and effects modeling (Fosberg and Deeming 1971, Rothermel 1972). On each transect, we tallied 1-h fuel from 10.3 to 12.1 m, 10-h fuel from 9.1 to 12.1 m, 100-h fuel and 1,000-r fuel from 0 to 12.1 m. For each 1000-h fuel intersection, we recorded species, DBH (cm), decay class (sound and rotten), and distance from plot center. CWD decay classes were classified into a five-class system of log decomposition which ranged from sound to rotten and based on characteristics described by Maser et al. (1979). Total length of Brown's (1974) transect for each 2.2 ha experimental unit were 176 m, 293 m, 1170 m, and 1170 m for 1-h, 10-h, 100-h, and 1000-h fuels; respectively. We used equations from Brown (1974) to convert numbers of intersections of dead woody fuels to fuel mass at each site. We used FFE-FVS to calculate average stand metrics (basal area, tree density) for each experimental unit and to simulate dead woody fuel succession and future fire behavior (reburn severity).

Monitoring plots including seedling and vegetation transects were established in 2016 to assess post-fire conditions. In each experimental unit, tree seedlings, shrubs, graminoids, and forbs data were collected on six 100 m transects from 2.25 m^2 quadrats installed every 5 m (20 per transect) extending across the plot centers (Fig. 2). Six 100 m transects were aligned with a 6 × 4 grid of inventory plots within each experimental unit. Quadrats were placed along the transect by aligning the left side of the square at 0 m (so the transect bisects the square), and then at 5 m intervals thereafter. Within each 2.25 m^2 square, we counted and measured the mean height of live conifer seedlings and identified species when possible. Percent cover of graminoids, forbs, shrubs, and *Calamagrostis rubescens* (Buckley pinegrass) were recorded in seven categories: 1 = < 1%, 2 = 1–5%, 3 = 5–10, 4 = > 10–25%, 5 = > 25–50%, 6 = > 50–75%, and 7 = > 75–100%. Mean shrub height was recorded.

2.3. Empirical data analysis

In this GRBD with replication (3 blocks, 3 treatments, 24 replicates in each block/treatment combination), there are pre- and post-treatment measurements for each fuel type. The treatments are fixed effects, and the blocks are random effects. First, we assessed whether the units differ significantly in basal area or snag density before the post-salvage logging treatments, then we evaluated the change in dead woody fuel loading (Post-Pre). For each of these we used a mixed-model two-factor ANOVA with replication, where the F-test for the treatment effect was calculated as the mean squared treatment divided by the mean square interaction (Zar 2010). This ANOVA was performed separately for each of the fine woody fuel types (1-, 10-, 100-hr) and coarse (1000-hr sound). In the GRBD, relevant comparisons depend on whether the treatment X block interaction is significant. If the interaction is not significant, then the treatment effect is the same regardless of block and the treatments can be summarized across the blocks. If the interaction is statistically significant, then the treatment effect differs among the blocks. This prevents general summaries across the blocks and requires the treatments to be compared within each block.

Post-hoc analysis was performed using Tukey's honestly significant difference (HSD), using the R function TukeyHSD (R Core Team 2013). If the treatment X block interaction was significant, HSD was performed within each treatment/block combination and all comparisons among treatments were visualized within each block. If the treatment X block interaction was not significant, HSD was performed with treatments aggregated by block, and all comparisons among treatments were summarized across the blocks. Boxplots of the fuel loadings showed no substantial departures from the normality assumption (Fig. S1), although the control treatments did exhibit substantially lower variability than the logged treatments. We decided to proceed with the ANOVA because the procedure was robust to non-constant variance in the case of balanced design. There were very few plots with larger fuels, or rotten fuels, in our measurement area. For those fuel types we only report summary statistics and do not perform statistical analysis.

2.4. Tree seedling and shrub data

To test whether post-logging seedling counts differ among the three treatments we summed seedling counts across all plots in a single transect, and use individual transects as a random variable in a generalized linear model with random effects and Poisson errors (R function glmer in the lme4 library; Bates et al. 2015). We performed an exploratory data analysis on percentage shrub cover data, as the data are not strictly numeric because we have the mean of the midpoints of range for cover. Therefore, for shrub cover data our analysis will be descriptive rather than inferential. We summarized mean midpoint shrub cover for each block and treatment combination both before and after the salvage logging treatment, as well as the change in cover pre and post-treatment.

2.5. Fuel succession and fire simulation modeling

To predict future fuel succession and to perform a simulation model test of the re-burn hypothesis, we used the Inland Empire Northern Idaho variant FFE-FVS (Rebain 2010; Keyser, 2008). FFE-FVS is a semi-empirical, distance-independent individual tree growth-and-yield models with region-specific allometric growth equations which simulates forest succession (tree growth and mortality), snag decay and fall down and dead woody fuel decomposition (Reinhardt and Crookston 2003). For all simulations we ran scenarios with 1680 trees ha^{-1} and 420 trees ha^{-1} planted in 2018 in each experimental unit to simulate artificial reforestation and projected each experimental unit 80 years.

For each experimental unit, we developed a fuel succession matrix to model short-and-long term dead woody fuel succession following harvest and tree planting. We built the matrix using FFE-FVS default snag fall down rates and dead woody fuel decomposition rates. To evaluate the potential impact of on our projections model uncertainty with respect to fuel decomposition and snag fall rates, we conducted simulations at both default values and at values +/-50% of default. To isolate treatment effects on future fuel accumulation we conducted one set of simulations without wildfire. We then compared CWD and FWD over simulated years among the three treatments.

To test the reburn hypothesis we conducted a second set of simulations that included a wildfire in the year 2037 (20 years after treatment). Our assumption is that the treated areas could experience a repeat wildfire within this time period. The wildfire was simulated with the 90th percentile (Lane Creek Remote Automated Weather Station) fire weather inputs: 1-hr fuel moisture (%): 4, 10-hr fuel moisture (%): 4, 100-hr fuel moisture (%): 5, 1000-hr fuel moisture (%): 10, duff moisture (%): 15, live woody moisture (%): 75, moisture for live herbs (%): 70, temperature (°F): 90, and windspeed (km/h): 32. FFE-FVS fire behavior projections were generated using Scott and Burgan's (2005) 40 fuel models.

We then compared FFE-FVS predicted fire behavior indices among the three treatments for each block. FFE-FVS fire behavior indices are (1) active crown fire (entire stand crown is involved in flame, but the crowning phase remains dependent on heat release from surface fuel for continued spread (Scott and Reinhardt 2001); (2) passive crown fire (wildfire burns crown of individual trees or groups of trees crowns; but solid flaming in the canopy cannot be maintained except for short periods (Scott and Reinhardt 2001); (3) surface (wildfire does not burn crowns of trees) and (4) conditional crown fire in which stand conditions support active crowning but do not support torching. For example, if the wildfire begins as a surface fire then it is expected to remain so. If the wildfire begins as an active crown fire in an adjacent stand; the wildfire will continue to spread as an active crown fire (Scott and Reinhardt 2001).

3. Results

3.1. Pre-salvage logging (2017 measurements)

Pre-treatment analysis found no significant differences among the treatments in stand basal area, stem density, tree species composition, or for any component of the fuelbed. Blocks did differ significantly in stem density, although they did not differ in basal area. Pretreatment average basal area was 27.6 m^2/ha. There were early seral stands with average stand height of 16.9 m. Tree density average was 1027 trees/ha with an average of 60.2%, 28%, 11.8% of trees were 0–15 cm, 15–30 cm, and > 30 cm, respectively. Average total woody fuel loading was 10.2 Mg /ha^{-1}. Across all blocks, pre-treatment total 1-hr fuel loadings ranged from 0.04 to 0.11 Mg ha^{-1}. Tree species distribution and composition varied by block. *Pinus ponderosa* and *Pseudotsuga menziesii* were dominant tree species.

3.2. Post-fire logging effects (1-year post; 2018 measurements)

Salvage logging significantly altered forest stand structural metrics and dead woody fuel loadings. For the most part, the initial specifications for the thinning prescriptions (SSR: thin to 3.4 m^2/ha; GTR: thin to 10.3 m^2/ha) were not accomplished in our experimental units, and snag basal area and snag density varied by treatment and block. Post-harvest residual BA for the SSR treatment units were 5.9, 8.8, and 4.1 m^2/ha for B1, B2, and B3 (the closest to the target prescription), respectively. Similarly, post-harvest residual BA for the GTR experimental units were 15.9, 6.5, and 8.7 m^2/ha for B1, B2, and B3, respectively. The SSR removed on average 79% of pre-treatment basal area compared to 62% reduction for GTR. Most harvested snags were small and unmerchantable which may have contributed to the variable implementation of harvest prescriptions.

Salvage logging increased dead woody fuel loadings but generated a wide range of results for the change in FWD and CWD loadings. Total woody fuel loading increased 80–106% (12.64–24.23 Mg ha^{-1}) in B1, 135–143% (13.76–24.77 Mg ha^{-1}) in B2, and 49–233% (15.11–38.85 Mg ha^{-1}) and in B3. The 1-hr fuel loadings were low (0.52 Mg ha^{-1}) despite a 372% increase. The SSR increased average CWD fuel loading above Brown's *et al.* (2003) lower recommended loadings of 11 Mg ha^{-1} (Fig. S2).

There were mixed results for the change in dead woody fuel loading depending on fuel size class (Fig. 4). The 1-, 10-, and 100-hr fuels had significant treatment and block X treatment effects (Table 1). The 1000-hr sound class did not have a significant treatment effect but did have a significant interaction. All fuel loadings tended to increase post-treatment in logged areas, whereas loadings tended to marginally decrease or not change in the control (unlogged) areas (Fig. 5; Table 1). With the block X treatment interactions, for all fuel loadings except 100-hr, we considered treatment effects individually for each Block. For 1- and 10-hr fuels, there was no significant post-hoc pairwise differences between treatments SSR and GTR, regardless of Block. For 100-hr fuel loadings, across all blocks, SSR had a significantly higher increase in total fuel loading than GTR. In comparing each treatment to the control, for 1-hr fuel loadings, treatment SSR had a significantly larger increase in dead woody fuels than the control in B1, with no significant increases in B2 and B3. Also for 1-hr fuel loadings, treatment GTR had a significantly larger increase in dead woody fuels than the control in B1 and B2, with no significant increases in B3. For 10- and 100-hr fuels, both salvage prescriptions resulted in significantly larger increases in fuel loading than the control, regardless of Block. With the significant interaction for 1000-hr sound, we found a significant increase in mean loading in B1 and B2 between treatment SSR and the control, with no significant increases between treatment GTR and the control. Only in B2 did we see a significant increase in mean 1000-hr sound loading between treatment SSR and GTR.

Table 1
Results of generalized randomized block design (GRBD) F-test for random block X treatment interaction and treatment main effect, for each fuel size. The value of the F-statistic is given, with the p-value in parentheses. Results of Tukey's honestly significant difference (HSD) post-hoc pairwise comparisons are also given, either for each block (B1, B2, B3) if the block X treatment interaction is significant, or across all blocks if the interaction is not significant (for 100-hr fuels). For example, in 1 v. control for 1-hr fuels the comparison is significant for B1 (B1***), but not for the other two blocks.

Response Variable	Block X Trt F (p)	Trt main F (p)	HSD SSR v. C	HSD GTR v. C	HSD GTR v. SSR
1-hr	5.44 (< 0.001)	7.62 (0.04)	B1*** B2 ns B3 ns	B1*** B2*** B3 ns	B1 ns B2 ns B3 ns
10-hr	3.02 (0.02)	23.6 (0.006)	B1*** B2*** B3**	B1*** B2*** B3***	nB1 ns B2 ns B3 ns
100-hr	1.65 (0.164)	43.3 (0.002)	all blocks ***	all blocks ***	all blocks ***
1000-hr Sound	9.37 (< 0.001)	2.08 (0.24)	B1*** B2*** B3 ns	B1 ns B2 ns B3 ns	B1 ns B2*** B3 ns

*** < 0.001; ** < 0.01; * < 0.05; ns means not significant.

3.3. Dead woody fuel succession

We limit FFE-FVS simulation results to the default decomposition and snag fall down rates. We report additional simulation results in supplementary material (Fig. S3). FFE-FVS predicted FWD and CWD woody loadings initially increase rapidly over time and then decrease (Fig. 5), indicating rapid stand collapse or surface fuel loading accumulation. Regardless of block and FFE-FVS model parameters, over time, CWD accumulation in the controls exceeded total accumulation in both treatments (Fig. 5, S3). In B1 and B2 this occurred in FVS year 2032 (15 years post-fire), and in B3 this occurred in FVS year 2036 (19 years post-fire). The total loadings of FWD remained relatively low over time (Fig. 5).

The comparison between treatments SSR and GTR depends on Block. In B1, the GTR treatment was predicted to gain more total CWD than SSR, whereas in B2 and B3 GTR accumulated lower total CWD over time than the SSR (Fig. 5). This corresponds to the post-treatment variability among the blocks. In B1 post-treatment snag BA was higher in GTR than in SSR, whereas in B2 it was lower and in B3 they were similar. Relative comparisons among the treatments do not vary with values of fall and decomposition rates, although the predicted values themselves do vary with snag fall rate and woody decomposition rate (Fig. S3).

3.4. Fuel and fire simulation (reburn)

The different planting densities affected forest structure metrics (canopy bulk density and canopy base height) which influence crown fire initiation and wildfire behavior. In 2036, average canopy bulk density was 0.04 kg m^{-3} for the higher planting density and 0.01 kg m^{-3} for the lower. The average canopy base height of the stands with the higher planting was 0.61 m compared to 1.52 m for lower planting density. The projected fire types between the two planting densities were similar. Forty-four percent of the experimental units were projected as passive crown fire regardless of planting density and fire type decreased from passive crown fire to surface fire in 22% of the stands between the high and low replanting densities.

3.5. Seedlings and shrubs

At $\alpha = 0.05$, there were no significant differences between each treatment and the control in the post-treatment seedling count ($p = 0.07$, 0.5 for treatment SSR and GTR, respectively), although the estimated effects are negative for both treatments relative to the control. Almost all blocks showed a decrease in seedling counts pre and post-logging, including the control plots (Fig. 6). There was substantial block-level variability in both the number of seedlings and in the change in seedlings. In B1, the SSR showed an increase in the number of seedlings (Fig. 6), whereas there was a decrease in both the GTR and in the control. B2 had the most seedling regeneration, with seedling counts orders of magnitude higher than either of the other blocks (Fig. 6). In B2, both treatments reduced seedling counts, whereas seedling counts increased in the controls. In B3, the SSR and the control had reduced seedling counts, whereas the GTR had a slight increase in seedling counts. Percent change in seedlings show remarkable consistency in magnitude, with the exception of the B1 intense logging treatment, with a massive percent increase in seedling count (Fig. 6). Percentage shrub cover, as measured by the mean midpoint in cover, increased in all treatments (including the control) and blocks pre to post-treatment (Fig. 7). Percentage shrub cover increased the highest in B1 and B2 controls, whereas in B3, shrub cover increased the highest in treatment SSR. There were no obvious consistent treatment effects on percentage shrub cover given these data and our descriptive analysis.

4. Discussion

The consequences of post-fire management on future fire hazard and woody fuel accumulations depends on the time since harvest or disturbance. Over immediate time scales (within years of the management action) woody fuel loadings increased with more intense post-fire logging. However, over time, fine woody fuel increases rapidly and then decrease as a result of decomposition. On the other hand, coarse woody fuel increases significantly providing conditions which could increase the severity of repeat wildfires. Effects on seedlings and shrubs were more ambiguous, with no statistically discernable effects.

4.1. Dead woody fuels: first-year post-harvest

As hypothesized, the extraction of fire-killed trees reduced average snag density, average snag basal area, and increased fine and coarse woody fuels loadings relative to control units (Fig. 3). In general, FWD and CWD tended to increase immediately post-treatment in logged areas relative to the unlogged controls (Fig. 4). This response is consistent with other studies showing fuel loadings tend to increase with timber harvest in the short-term as fire killed trees deposit woody fuel to the ground (McIver and Ottmar 2007; Keyser et al. 2009; Peterson et al. 2015). This production of slash (unmerchantable biomass including tree branches and tree-tops; Donato et al. 2006), which tends to increase short-term fire hazard potential (McIver and Ottmar 2007). Fuel loadings measured here are consistent with McIver and Ottmar (2007) and Keyser et al. (2009), who both had similar pre-harvest average stand basal areas to our units. For example, Keyser et al. (2009) recorded maximum fuel loadings (1 and 5 years post logging) of 12.8 Mg ha^{-1} and 29.5 Mg ha^{-1}, respectively. McIver and Ottmar (2007) measured maximum loading for treatments of 16.9 Mg ha^{-1}. In all of these studies, the short-term post-salvage logging fuel loadings did not exceed the upper threshold of 45 Mg ha^{-1} that Brown et al. (2003) recommended for maintaining long-term site productivity and other ecosystem services (wildlife habitat) for warm-dry forest types. Logging operations, specifically whole-tree harvesting, probably was the key to controlling the transfer of crown material (branches) to the forest floor, therefore limiting subsequent fuel loading accumulations (Donato et al. 2006; McIver and Ottmar 2007; Keyser et al. 2009; Peterson et al. 2015; Ritchie et al. 2013). For comparison, surface fuel loadings increased significantly on the Biscuit fire because snags were hand felled and de-limbed on site before being extracted via helicopter or cable logging (Donato et al. 2013). Donato et al. (2006) considered mean fuel loadings of 6.7 Mg ha^{-1} a significant fire hazard for early seral stands. In our experimental units, we found no evidence that post-fire salvage logging necessitates subsequent fuel treatments (prescribed fire or pile and burn) to reduce elevated woody fuel loadings (Donato et al. 2013). Although fuel loadings are within Brown's (1974) threshold, fire and forest managers may decide to implement post-harvest fuel treatments (prescribe burns or pile and burn) to increase forest resilience for the regenerating coniferous forest and to improve fuelbed conditions for reforestation activities (Agee and Skinner 2005).

When we compared the two salvage logging treatments we did not, however, find significant differences between the two treatment intensities for 1-hr and 10-hr fuels (Table 1), whereas the 100-hr fuels did increase significantly more for the more intense logging treatment (SSR). The substantial variability in the implementation of the salvage logging treatments (Fig. 3). The block-level variability may be related to inconsistency in the on-the-ground implementation of the logging treatments or differences in stand environmental or biophysical conditions (biophysical features (e.g., topography, vegetation composition and structure, fuel conditions). The significant interaction between treatment effect and block requires evaluating post-treatment stand structure and fuel accumulations separately for each block. Our study results show the initial (short-term) fuel loadings and long-term fuel accumulations are proportional to harvest prescription (Donato et al. 2013; Ritchie et al. 2013). Where significant differences between the two thinning intensities occurred, the SSR (intense logging prescription) produced a larger mean increase in woody fuels (particularly for 100-hr fuels; Fig. 4). These results show that stand density and stand basal area as good predictors of bole and branch biomass (Donato et al. 2013, Ritchie et al. 2013), perhaps better predictors than a given treatment prescription classification.

Post-treatment fine woody fuel loadings (Fig. 5) were inconsequential to model-predicted wildfire behavior metrics (rate-of-spread, flame length) important to fire managers. These metrics are calculated from the loading of 1-hr fuels (Rothermel 1972), but post-harvest CWD fuel loading can produce undesirable fire behavior effects such as increase flame residence time, smoldering combustion, radiant heat fluxes (Sullivan et al. 2002) and prolonged soil heating and total heat release (Brown et al. 2003; Monsanto and Agee 2008). CWD loadings do not influence wildfire rate-of-spread, fireline intensity and flame lengths (DeBano et al. 1998; Hyde et al. 2011).

4.2. Post-fire dead woody fuel succession

FFE-FVS simulation modeling show a rapid accumulation of dead woody fuels (initial pulse period 0–25 years; Fig. 5), and these generated outputs are consistent with recent salvage logging fuel loading succession studies and empirical studies that found snag fall and fuel accumulation within the first decade following wildfire (Keyser et al. 2009; Ritchie et al. 2013; Dunn and Bailey 2012; Peterson et al. 2015). Several studies have concluded that woody fuel loadings in control stands exceeded woody fuel loadings in logged stands within 4–15 years (Donato et al. 2006; McIver and Ottmar 2007; Monsanto and Agee 2008; Keyser et al. 2009; Peterson et al. 2015).

In our analysis the FWD peaked earlier than CWD, within 10 years from time since wildfire and all before our simulated wildfire in 2037. Empirical data and simulation modeling show elevated fuel loading up to 20 years post-treatment (McIver and Ottmar 2007; McGinnis et al. 2010) and this time period may correspond to the highest wildfire risk. In our study, CWD fuel loadings exceeded Brown's (2003) threshold within 19 years (B3) and 15 years (B1 and B2) of post-fire management, and then decreased gradually over time. Low residual snags in logged units reduces the amount of crown material to be transferred to the surface fuel pool (McIver and Ottmar 2007).

4.3. Reburn hypothesis

Severity of reburns of future wildfires is a central forest management concern (Coppoletta et al. 2016). Post wildfire snag succession and regenerating nonwoody vegetation (shrubs) can affect future wildfire behavior (reburn) (McIver and Ottmar 2007, Coppoletta et al. 2016, Harris and Taylor 2017). A central question is what factors influence reburn severity and can reburn severity be mitigated with postfire salvage logging. Reburn severity is a function of the availability (loading or quantity) of both live vegetation and dead woody fuel loadings (Collins et al. 2009). In general, fire severity in repeat wildfires tend to mirror the previous burn severity (Thompson et al. 2007; Coppoletta et al. 2016; Harris and Taylor 2017). Several studies have concluded areas that burned at high severity have a higher probability

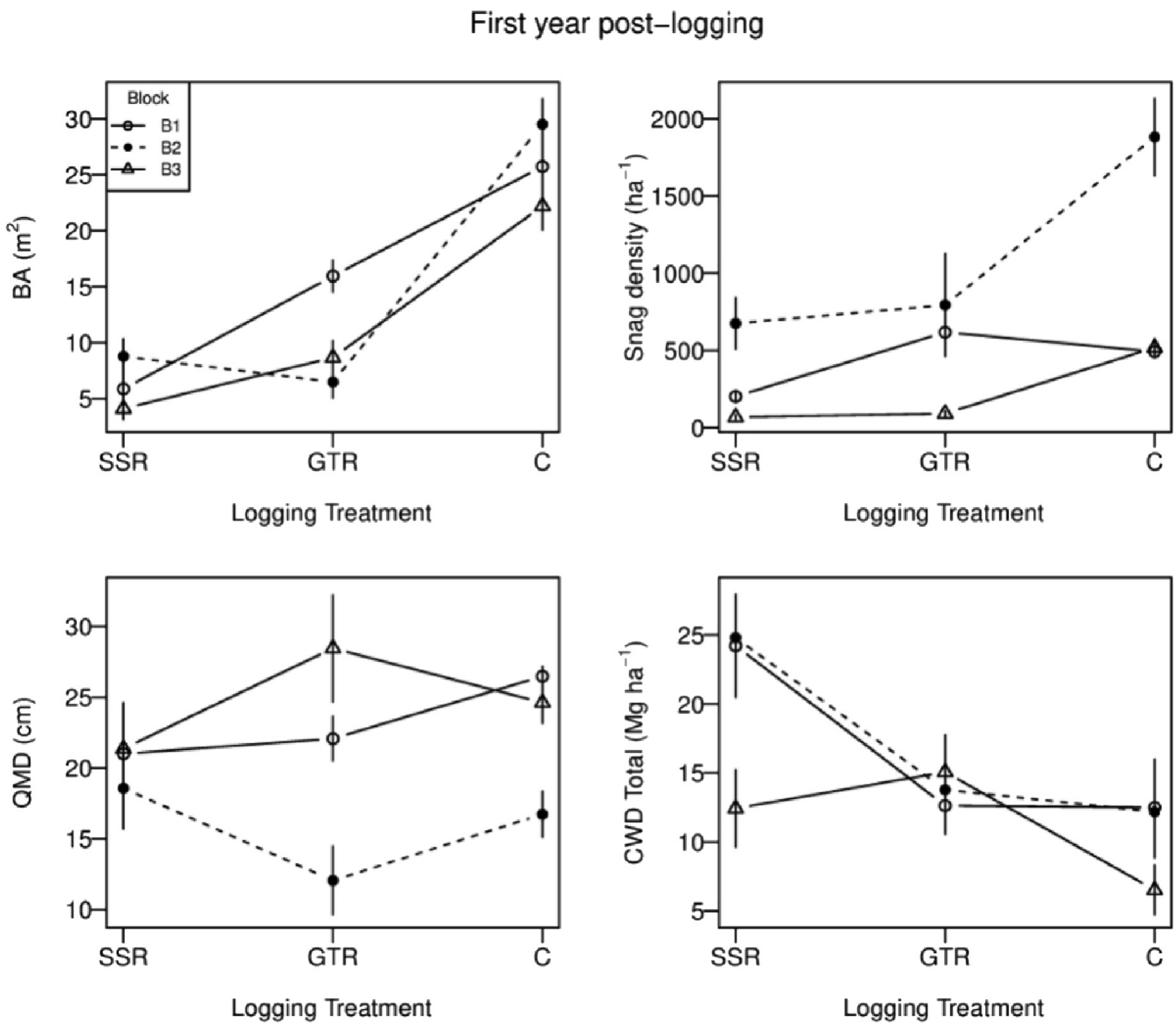

Fig. 3. Snag metrics and CWD following salvage logging operations on the Stickpin wildfire. Stickpin thinning prescriptions were thin to 3.4 m^2/ha (SSR) and 10.3 m^2/ha (GTR). Compared to the arithmetic mean, QMD assigns greater weight to larger trees – QMD is always greater than or equal to arithmetic mean for a given set of trees.

to reburn at the same severity (Thompson et al. 2007; Holden et al. 2010; van Wagtendonk et al. 2012; Coppoletta et al. 2016). Passive management following stand-replacing wildfires can perpetuate successive high-severity reburns (Thompson et al. 2007). Our FFE-FVS simulations provide limited evidence on reburn severity, particularly in the control units, as dead woody fuel accumulated above Brown et al. (2003) CWD thresholds. Time since disturbance (wildfire) and capacity for shrub and herbaceous biomass to reestablish influences reburn severity (Thompson et al. 2007). However, FFE-FVS does not model the succession of shrub and herbaceous biomass and therefore limit our predictions on future reburn severity.

Post-disturbance reforestation practices designed to expedite conifer forest rehabilitation can influence reburn severity and forest resilience (Thompson et al. 2007; McIver and Ottmar 2007). Artificial regeneration (seedling plantings) is a common forest management practice to control stocking levels and species composition following large-scale disturbances (North et al. 2019). Historically, these areas were replanted to dense plantations (high tree density). In the short-term (< 15 years), forest plantations are not resilient to repeat wildfires (Thompson et al. 2007) as reburns in early-seral stands cause high burn severity or tree mortality (Thompson and Spies 2010; van Wagtendonk et al. 2012). At the time of our simulated wildfire, the treatment units were ~19 year-old plantations with an average of 998 trees/ha^{-1}. The average canopy bulk density (mass of available canopy fuel per unit canopy volume) was 0.04 kg m^{-3} and an average canopy base height (lowest height above ground above which there is sufficient canopy fuel to propagate fire vertically) was 0.61 m. High canopy bulk density and lower canopy base height are associated with crown fire initiation and propagation (Van Wagner 1977; Agee and Skinner 2005; Peterson et al. 2005). The FFE-FVS model predicted passive crown fire (crowns of single trees or small groups of trees burn) for most treatment units.

The 2037 simulated wildfire exhibited high severity characteristics by decreasing total woody fuel loadings and killed on average 89% of the planted seedlings. McIver and Ottmar (2007) concluded high reburn severity is inevitable regardless of salvage logging activities as early seral stands (stand initiation stage) are susceptible to active crown fire. Campbell et al. (2016) found live sclerophyllous vegetation dominate early seral forests within seven years after logging, leading to a convergence in fire severity risk among experimental treatments. Replanting to lower stem densities or wider spacing can reduce the development of closed-canopy, homogeneous young stands which are susceptible to crown fire initiation. New plantations would require active management to maintain low fire hazard. Precommerical thinning 10–15 years after planting would be necessary to increase forest resilience and reduce crown fire hazard (Agee and Skinner 2005, Peterson et al. 2005). Altering standard reforestation practices using methods such as variable-density reforestation can increase forest heterogeneity and reduce reburn severity (Larson and Churchill 2012).

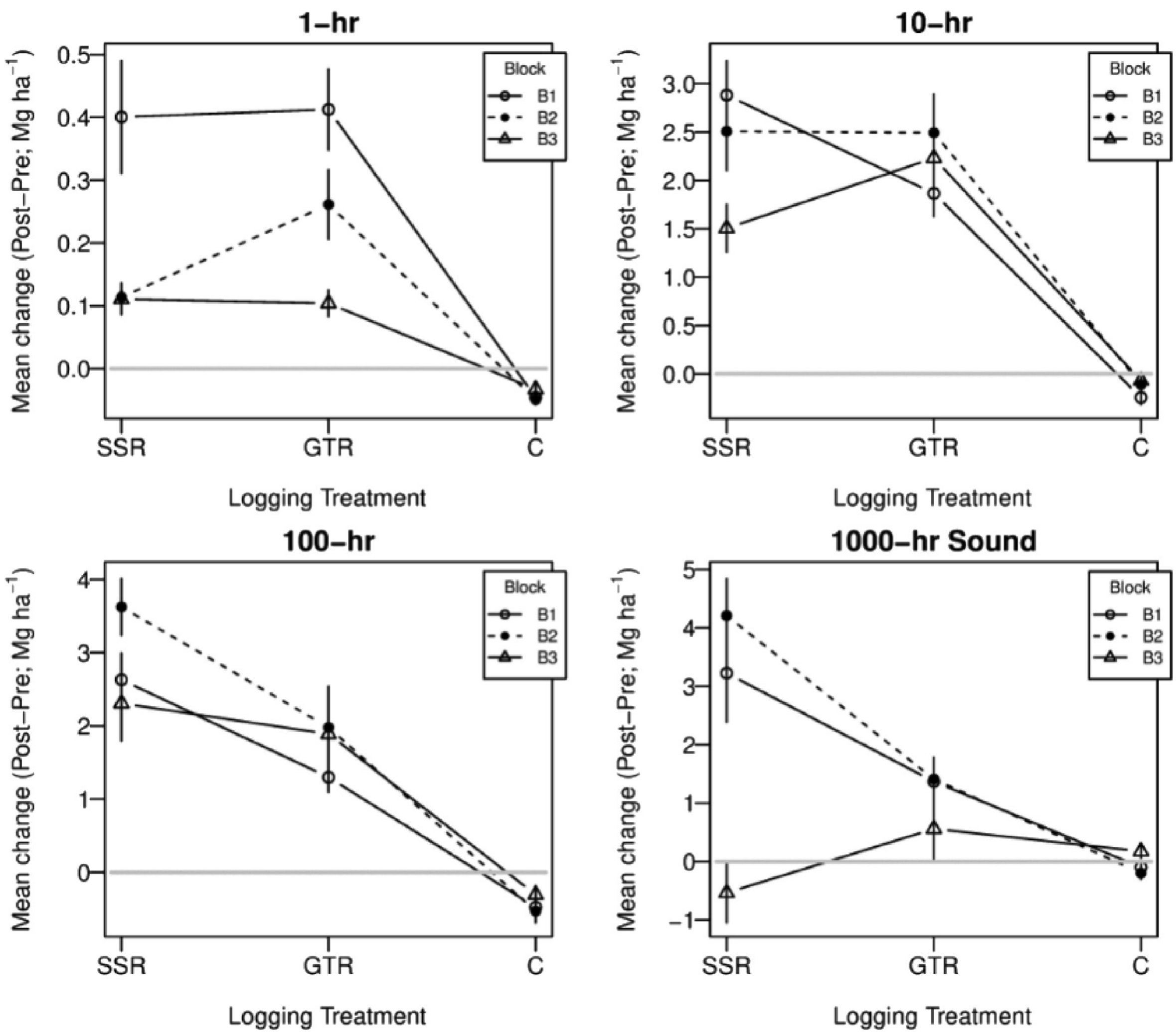

Fig. 4. Observed mean change in fuel loading (Post-Pre-treatment), for each block and salvage-logging treatment for four fuel size classes. Vertical lines represent +/1 1 standard error.

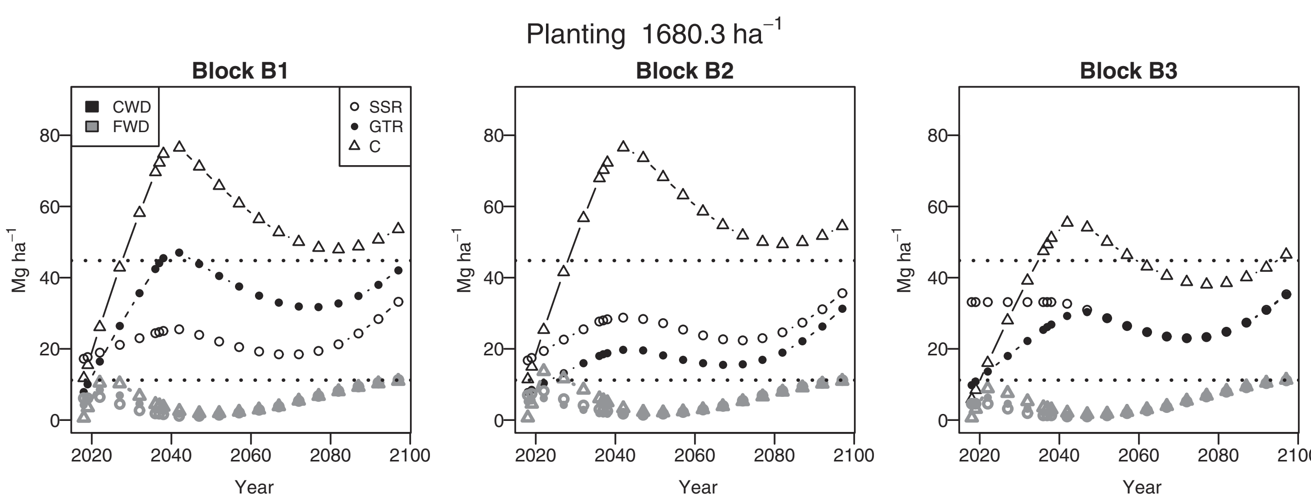

Fig. 5. Fuel succession (CWD and FWD over time), by treatment and block (no simulated wildfire). Grey horizontal line at 44.8 Mg ha^{-1} and 11.2 Mg ha^{-1} represent optimum ranges of coarse and fine woody debris, respectively.

Following large stand-replacing wildfires, which produce large mosaics of contiguous mortality areas, North et al. (2019) proposed a three-zone reforestation plan as an alternative to traditional dense even-spacing planting specifications. However, forest managers must consider the tradeoffs between alternative reforestation practices, as these adaptive management strategies can promote rapid shrub regeneration, which can increase moisture deficit, increase fire hazard, reduced seedlings, decrease the success rate of forest reforestation (North et al. 2019).

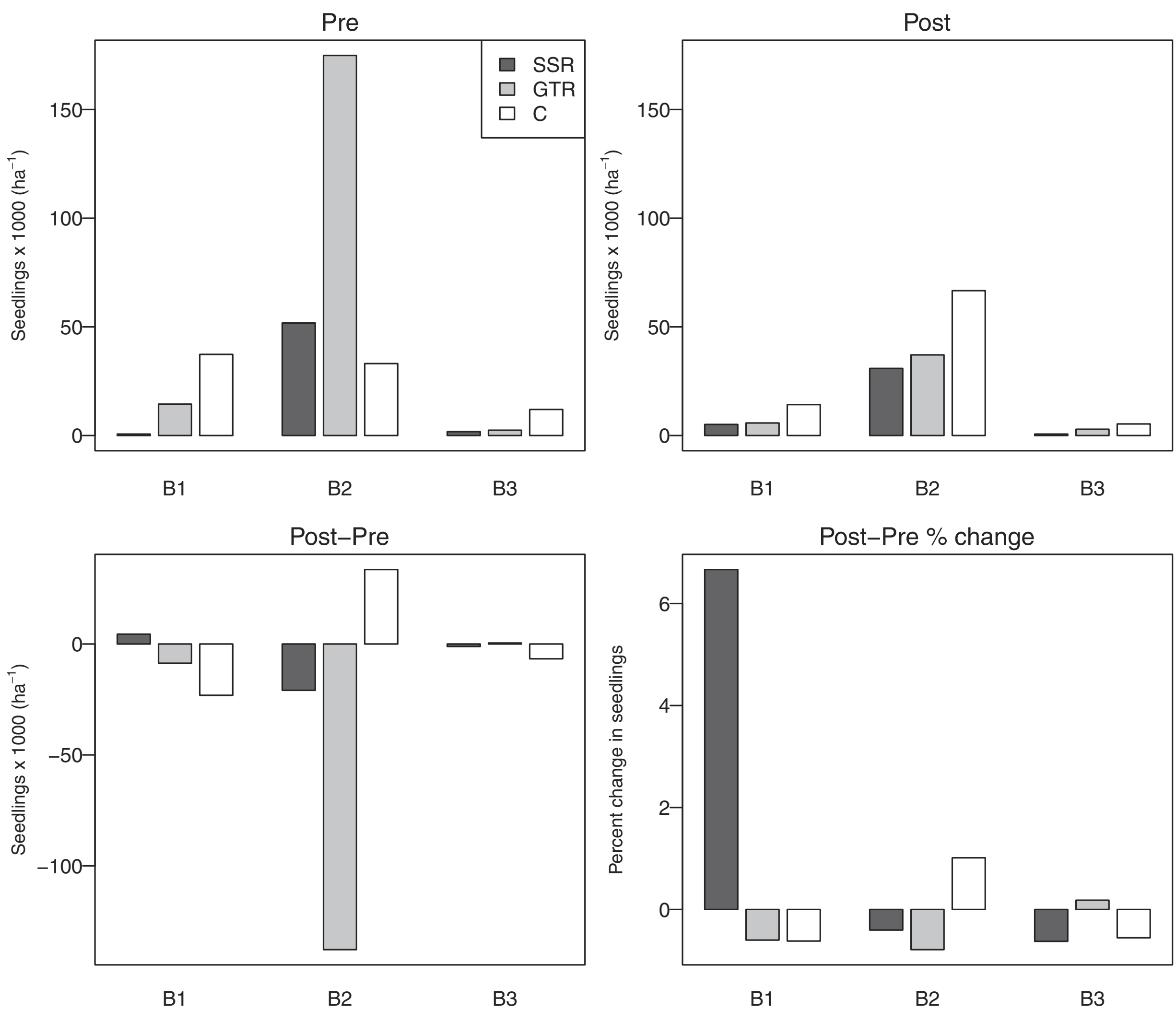

Fig. 6. Seedling density (a) before treatment, (b) after treatment, (c) difference, and (d) percent difference.

Lower seedling density or wider planting spacing can reduce future crown fire hazard but also encourages prolific shrub/grass regeneration in the available growing space which could increase crown fire initiation (McDonald and Fiddler 2010).

4.4. Seedling survival and shrub coverage

Salvage logging effects on seedlings and shrubs were mixed (Fig. 7). We found no significant difference between each treatment and control. For seedlings the estimated effects were negative for both treatment relative to the control (Table 2). We recorded the largest seedling change in B2 (SSR) and control units (Fig. 7). Our measured response reflects seedling counts 1 year after salvage logging and seedling density may increase in subsequent years. Overall, harvest effects on seedling density and shrub coverage were consistent with other research studies (Keyser et al. 2009; Peterson et al. 2015). Seedling mortality is inevitable during forest logging operations as mechanical harvesting equipment crush and bury pre-harvest seedling and shrubs (Fernández et al. 2008). Post-logging seedling mortality and increase in woody fuel loadings is probably common with both green tree harvest and salvage logging, and seedling reduction is undoubtedly related to harvest method and degree of ground disturbance (McIver and Starr 2001; Wagenbrenner et al., 2015). Overall seedling density decreased in most units regardless of whether the unit was subject to post-fire treatment. Massive block-level variability (and interactions) may mask any potential treatment effect on seedling survival. Previous studies show different logging effects on seedling survival (Donato et al. 2006), from no negative impact (Peterson and Dodson, 2016) to significant short-term negative impacts (Sexton 1998). However, natural regeneration survival probably is not a management concern because these areas would be manually replanted to accomplish policy reforestation mandates (NFMA 1976).

There is a correspondence between density of snags (as B2 had the highest density) and seedling which may be related to the biophysical attributes of B2 relative to other blocks. Higher density of seedlings also corresponds to higher magnitudes of change in seedlings. It also seems that the effect of salvage logging on seedling density is minimal when there is little regeneration, and there is preliminary evidence that following salvage logging it is still possible for seedlings to regenerate naturally as salvage logging disturbance provide favorable conditions for seedling establishment (Peterson and Leach 2008). So, when seedlings are abundant, (as in Donato et al. 2006), salvage logging appears to reduce the seedling counts or increase seedling mortality. When there are very few seedlings, salvage logging has less of an effect. Note that these results are ambiguous statistically, with mixed results in general and that we did have pre-treatment differences among the treated areas that further confound our interpretation of salvage logging effects on seedlings. If maintaining natural regeneration is considered an objective, when choosing areas for logging, avoid areas with evidence of high-density seedling regeneration.

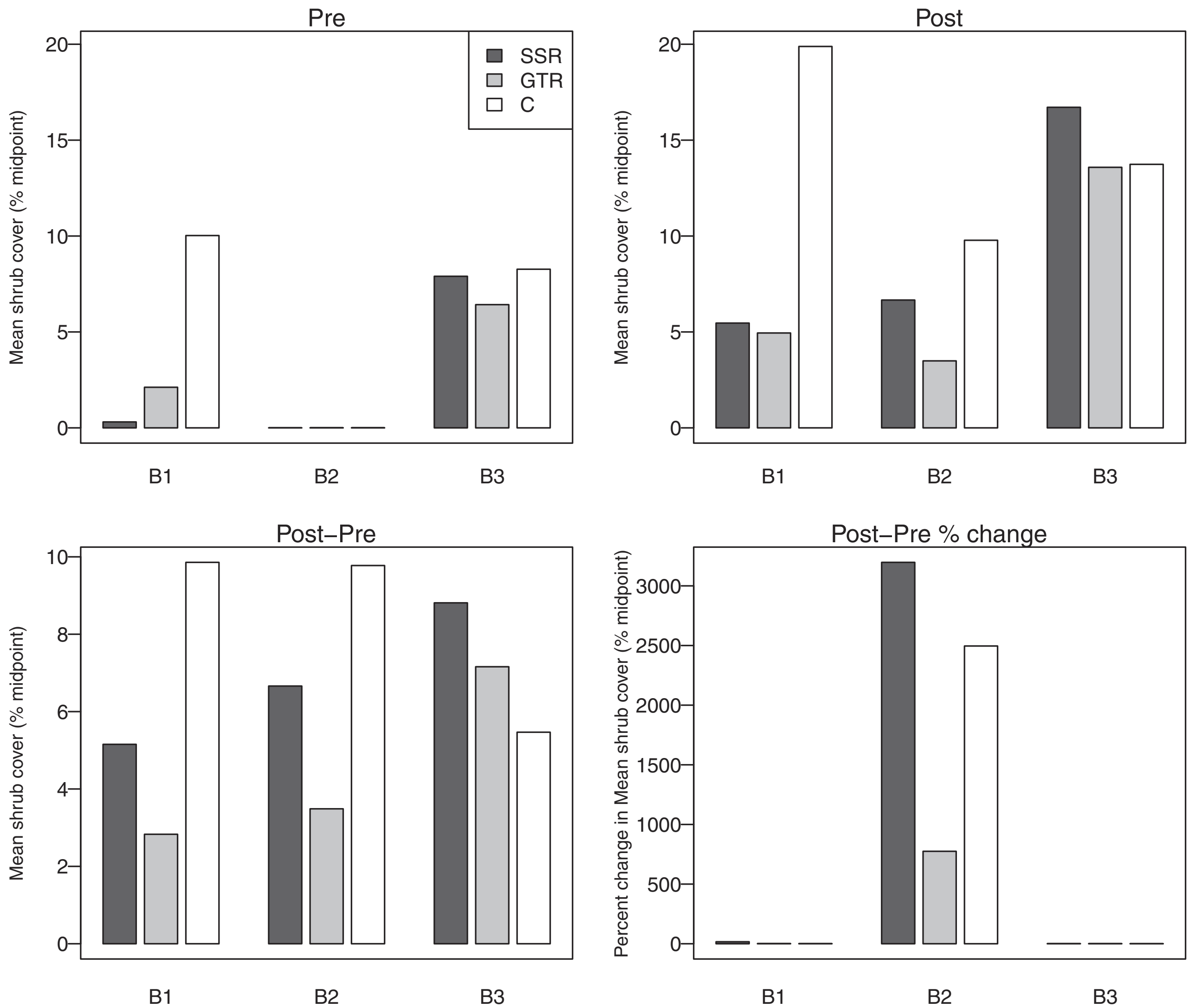

Fig. 7. Pre and Post-treatment shrub percent cover (midpoints), change in shrub percent cover (post-pre) and percent change in shrub percent cover.

Table 2
Estimated model for post-treatment seedling counts (fixed effects only). Note that all coefficients indicate the change in log seedling count relative to the control.

Coefficient	Estimate	Standard error	z (p)
Intercept	1.83	0.41	4.44 (< 0.001)
SSR	−1.10	0.62	−1.77 (0.07)
GTR	−0.34	0.59	0.58 (0.56)

In general shrub cover increased in the year after the logging treatment across all units including the control (Fig. 7). The comparisons among the treatments differed by block. The block with the lowest pre-treatment shrub cover (block 2) had the largest increase in shrub cover both in magnitude and on a percentage basis. In blocks 1 and 2 the control had the largest magnitude increase in shrub cover, whereas in block 3 the SSR had the largest magnitude increase in shrub cover. Given these data it is not possible to make generalizations about the impact of salvage logging on shrub cover. Further research in this area is warranted.

5. Conclusions

Given predictions of a warming climate (Haire and McGarigal 2008) and increased wildfire activity (Littell et al. 2009), understanding the fuel trajectory of large mosaics of fire-killed trees is essential for forest managers. With a backlog of millions of hectares of overcrowded, dead and dying, dry forests requiring hazardous fuel treatments (Agee and Skinner 2005), large stand-replacing wildfires may be inevitable. As future wildfires produce large mosaics of dead trees these areas are placed on a successional pathway of repeat high severity wildfires unless there is fuel management intervention. Salvage logging fire-killed trees is a preemptive restoration treatment, and the decision to remove fire-killed trees following large wildfires is complicated and contentious (Beschta et al. 1995; Lindenmayer et al. 2008). Forest managers have to balance wildfire management goals and wildlife habitat requirements in post wildfire forests in order to make informed management decisions.

There are four main conclusions to this study:

1. Salvage logging is feasible without significantly increasing woody fuel loadings and wildfire hazard (Donato et al. 2006), particularly when whole-tree harvesting is used. This contributes to dry coniferous forest restoration objectives of reducing fuel loadings and potential fire hazard (Peterson et al. 2015).
2. Prescriptions can be developed that still retain higher levels of snags (< 10 m^2/ha) to balance multiple objectives, including maintenance of wildlife habitat. Prescriptions that seek to retain higher

levels of snags, such as those that mimic green-tree thinning treatments, appear to reduce fuel loadings to within acceptable levels while providing additional snags and future downed wood for wildfire habitat.
3. If lower fuel loadings are desired, additional fuels treatments are necessary beyond removal of merchantable trees.
4. The implementation of salvage logging treatments, as well as their effects, are highly variable even within the same management unit. Managers should consider inspecting fuel conditions post-logging and determine if a post-salvage fuels treatment is warranted to meet the long-term goals of the site. Methods to standardize treatment implementation should also be evaluated.

6. Limitations and suggestions for future research

These results must be considered in the context of using FFE-FVS model predictions. These predictions can help inform decisions on post-fire management directions with respect to potential future fire behavior and fuel loadings. However, our long-term projections on fuel succession and subsequent fire behavior were generated from a simulation model that has limitations; specifically, the fire behavior model outputs have considerable uncertainty (Cruz and Alexander 2010). FFE-FVS does not simulate the shrub and herbaceous regeneration which are major contributors to reburn severity and wildfire behavior (Rothermel 1972) and our sampling design does not consider dead woody fuel continuity which influence wildfire spread (Donato et al. 2006). However, fuel continuity probably is not an issue because skid trails likely increased fuel heterogeneity.

The conclusions are also limited in that the experiment is from one wildfire and management area. While the patterns observed here can inform general strategies for salvage logging in the context of unlogged controls, the actual estimates of post-logged fuel loadings and future fuel accumulations may not apply to different ecosystems under different wildfire and management conditions. However, overall, our general findings are consistent with a myriad of other salvage logging research studies (McIver and Ottmar 2018), which supports their robustness.

Given the limitations of model-predictions, it would be worthwhile to invest in empirical longitudinal studies of woody fuel accumulations with and without salvage-logging. These can help to corroborate model predictions, and provide empirical estimates of fuel succession. Furthermore, studies such as this are opportunistic by necessity, and each represents an individual independent contribution to the existing body of knowledge. Additional opportunistic studies would help to understand how the results can be extrapolated and to improve our ability to predict fuel accumulation. Furthermore, we demonstrate here that it is possible to design salvage-logging prescriptions of different intensities that may balance multiple management objectives. As these different prescriptions are implemented their consequences on woody fuels and other management objectives can be monitored and shared.

Given the variability we observed in treatment implementation and effects, future research studies should also focus more attention on marking harvest prescription and implementation. This would require more collaboration with the local silviculturist and timber sale administrator of the project.

CRediT authorship contribution statement

Morris Johnson: Conceptualization, Data curation, Formal analysis, Funding acquisition, Investigation, Methodology, Project administration, Writing - original draft, Writing - review & editing, Funding acquisition, Resources, Supervision. **Maureen Kennedy:** Conceptualization, Software, Data curation, Formal analysis, Funding acquisition, Investigation, Methodology, Project administration, Writing - original draft, Writing - review & editing, Formal analysis. **Sarah C. Harrison:** Software, Formal analysis, Investigation, Data curation, Visualization. **Derek Churchill:** . **James Pass:** Writing - review & editing. **Paul Fischer:** Writing - review & editing, Investigation.

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Appendix A. Supplementary material

Supplementary data to this article can be found online at https://doi.org/10.1016/j.foreco.2020.118190.

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