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Long-term effects of fuel treatments on aboveground biomass accumulation in ponderosa pine forests of the northern Rocky Mountains

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ABSTRACT

Fuel treatments in ponderosa pine forests of the northern Rocky Mountains are commonly used to modify fire behavior, but it is unclear how different fuel treatments impact the subsequent production and distribution of aboveground biomass, especially in the long term. This research evaluated aboveground biomass responses 23 years after treatment in two silvicultural installations with different cutting and underburning prescriptions in western Montana. The thinning installation included control (no treatment), thin/no burn, thin/spring burn, and thin/fall burn treatments. The shelterwood installation included control, cut/no burn, cut/wet burn, and cut/dry burn treatments. Across all fuel treatments in both the thinning and shelterwood installations, tree biomass had recovered to pre-harvest levels by 2015, or 23 years post-treatment. In the thinning, total aboveground and live-tree biomass were greatest in the control, but did not differ among the three thinned fuel treatments. Forest floor biomass was lower in the two burned treatments relative to the two unburned treatments. Seedling, vegetation, stump, and snag biomass did not differ among the four treatments. In the shelterwood, total aboveground and live-tree biomass were both greater in the unburned treatments relative to the burned treatments. Forest floor and snag biomass also tended to be lower in the burned treatments. Seedling, vegetation, and stump biomass were similar across all treatments. This research shows that tree biomass in ponderosa pine stands subjected to common fuels treatments can recover to pre-harvest levels in less than 23 years, while still exhibiting reduced stand densities that promote forest restoration objectives. Burgeoning biomass at the seedling layer suggests that additional understory treatments are necessary in order to abate ladder fuel development and sustain resistance to high-severity wildfire.

1. Introduction

In the western United States, many low-elevation, dry forest types have experienced substantial increases in forest biomass over the past century. For instance, between 1953 and 2012, net forest volume in the Intermountain West increased by 30 percent (Oswalt et al., 2014). This increase in biomass is primarily attributed to declines in logging activity and the rise of fire exclusion policies. Prior to European settlement, low-to-mixed severity fires burned western dry forest types at average intervals of 3–50 years (Hessburg and Agee, 2003; Fitzgerald, 2005). However, logging, grazing, and fire suppression have all contributed to interruptions in natural disturbance regimes (Pyne, 1982; Agee, 1993; Stephens and Ruth, 2005; van Wagtendonk, 2007; Naficy et al., 2010), with dry forest types burning less frequently than historically (Parks et al., 2015). It is estimated that of the 236 million acres in the West identified as forestlands, 67 million acres have been moderately or significantly altered by fire exclusion (USDA Forest Service, 2005).

Evaluation of carbon storage capacity in fire-prone landscapes is difficult because of the variable nature of fire and resulting effects (Loehman et al., 2014). An oft-cited inadvertent benefit of the suspension of natural disturbance regimes and increases in forest biomass has been a net gain in carbon sequestration throughout many of the dry forest types in the West, which has helped to offset fossil fuel emissions (Sohngen and Haynes, 1997; Houghton et al., 2000; Hurtt et al., 2002). However, as changing climate conditions result in longer and drier fire seasons, fire exclusion is becoming less feasible, and 100 years of fuel accumulation is resulting in fires of much higher severity, intensity, and magnitude than those historically present on the landscape (Westerling et al., 2006; Flannigan

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et al., 2013; van Mantgem et al., 2013). Carbon emitted from high-severity forest fires can be substantial, and in some years, even exceed regional annual carbon emissions from fossil fuels (Wiedinmyer and Neff, 2007; Dore et al., 2008). This transformation of a forest from a sink to a source is known as an offset reversal (Mignone et al., 2009). Recent studies suggest that fuel management treatments can prevent offset reversal in forests with low and mixed-severity fire regimes by reducing the high-severity wildfire hazard (Loehman et al., 2014).

Wildfire hazard is typically mitigated by applying a diverse set of silvicultural, mechanical, and prescribed fire strategies to alter the quantity and structure of forest fuel complexes, and thereby alter potential fire behavior (Graham et al., 2004). For instance, crown fire hazard is often addressed by reducing canopy density, removing small diameter ladder fuels, and increasing canopy base height (Graham et al., 1999; Keyes and O'Hara, 2002; Agee and Skinner, 2005; Fitzgerald, 2005; Reinhardt et al., 2008). Activity fuels produced by harvesting are typically treated by piling and burning, mastication or mulching, or broadcast burning (Agee and Skinner, 2005). While the effectiveness of these treatments on reducing subsequent fire severity is well documented (Kalies and Yocom Kent, 2016), uncertainties remain as to how these fuel treatments impact ecosystem biomass and whether or not the net amount of carbon released from fuel reduction treatments is less than that of potential wildfire emissions. There is increasing evidence that reducing wildfire severity, even at the cost of initial carbon reductions from thinning, will result in a more sustainable carbon sink over the long term (Hurteau et al., 2008; Hurteau and North, 2009; Amiro et al., 2010; Dore et al., 2010). For example, Hurteau et al. (2008) found that thinning prior to wildfire occurrence could reduce carbon emissions from live tree biomass by as much as 98%. As wildfire is an inevitability across much of the western landscape, implications of these treatments for offsetting carbon emissions may be substantial.

In addition to reducing emissions from high-severity wildfires, fuel treatments may help restore attributes of pre-settlement forest structure. There is evidence that the fewer large trees present in western forests prior to European settlement stored more carbon than the abundance of smaller trees that currently exist (Fellows and Goulden, 2008) and that large trees both fix and store more carbon relative to small trees (Stephenson et al., 2014). By reducing competition for water, nutrients, and light, residual trees may experience increased photosynthetic rates after harvest, increasing the amount of carbon sequestered per individual tree (Feeney et al., 1998; Skov et al., 2004; Sala et al., 2005; Dore et al., 2010). Increased vigor of individual trees may also contribute to maintaining long-term live biomass carbon sinks in the face of increasing occurrences of drought and other disturbances such as insects and disease (Larsson et al., 1983; Skov et al., 2004; Hood et al., 2016).

It is estimated that of the 6.9 billion bone dry tons of standing timber volume in the 15 western states of the United States, removal of approximately 2 billion bone dry tons (almost 30%) is required from forests with historically low and mixed-severity fire regimes in order to restore pre-settlement biomass quantities (USDA Forest Service, 2005). However, there is uncertainty regarding where and how this biomass should be removed to balance carbon storage objectives with ecological restoration objectives. Moreover, it is unclear how biomass will respond to these treatments in the long-term, as very few studies have examined biomass trends over periods longer than 3 years post-treatment.

This study evaluated aboveground biomass in a ponderosa pine (*Pinus ponderosa* Lawson & C. Lawson var. *ponderosa* C. Lawson)/Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco var. *glauca* (Beissn.)) forest in western Montana 23 years after application of several common fuel reduction treatments. Our primary objective was to determine how treatments impact each of the different components of aboveground biomass, and to describe the development of these components over the 23-year post-treatment period. We hypothesized that 23 years after treatment (1) live tree biomass would remain lower in all fuel reduction treatments relative to pre-treatment biomass levels, while (2) forest floor biomass and understory vegetation biomass would recover since time of treatment and be consistent across all treatments. We also predicted that (3) seedling biomass would be greatest in the thinning without prescribed fire treatments, and that (4) snag biomass would be greatest in the untreated control. Finally, we expected (5) total aboveground biomass in each of the fuel reduction treatments would continue to be lower than the control 23 years after treatment. This research is directed at helping managers anticipate the long-term effects of fuel reduction treatments on the structure of aboveground biomass in northern Rocky Mountain ponderosa pine forests.

2. Methods

2.1. Study site

Research was conducted at the Lick Creek Demonstration/Research Forest (hereafter: Lick Creek) on the Darby Ranger District of the Bitterroot National Forest in southwestern Montana (46°5′N, 114°15′W) (Fig. 1a). Forest management in portions of the Lick Creek drainage began in 1909, and the area has a long history of documented research studies (Smith and Arno, 1999). The site is semi-arid, with an estimated average annual temperature of 7 °C and precipitation of 400 mm, with about 30% of this annual precipitation falling as snow (Gruell et al., 1982; DeLuca and Zouhar, 2000). Elevations within Lick Creek range from approximately 1300 to 1500 m, with slopes primarily ranging from 0 to 30 percent (Menakis, 1994). Soils are relatively shallow or moderately deep, and are classified as Elkner Gravelly Loam, coarse-loamy, mixed, frigid Typic Cryochrepts, with highly weathered granite parent material (DeLuca and Zouhar, 2000).

Overstory vegetation consists principally of ponderosa pine and intermittent Douglas-fir, with grand fir (*Abies grandis* (Douglas ex D. Don) Lindl.), subalpine fir (*Abies lasiocarpa* (Hook.) Nutt. var. *lasiocarpa*), and lodgepole pine (*Pinus contorta* Douglas ex Loudon var. *latifolia* Engelm. ex S. Watson) occasionally present. Habitat types as classified by Pfister et al. (1997) within the drainage are Douglas-fir/snowberry (*Symphoricarpos albus* (L.) S.F. Blake) and Douglas-fir/pinegrass (*Calamagrostis rubescens* Buckley) located on the southerly aspects, and Douglas-fir/dwarf huckleberry (*Vaccinium caespitosum* Michx.), Douglas-fir/blue huckleberry (*Vaccinium globulare* Douglas ex Torr.), Douglas-fir/twinflower (*Linnaea borealis* L. subsp. *americana* (Forbes) Hultén ex R.T. Clausen) and grand fir/twinflower on the northwest aspects (Menakis, 1994).

2.2. Experimental design

Two installations were examined: a commercial thinning and a retention shelterwood (Table 1) that were concurrently established as independent studies, each with a complete block design and subsampling. Each installation has four treatments replicated three times for twelve experimental units, with 12 permanent plots per unit. The treated units were randomly assigned, but the control (no treatment) units had non-random placement due to logistical reasons for the prescribed burns. We refer to the control units as "untreated" and the harvested/burned units collectively as "treated." Non-permanent inventory plots measured prior to unit designation provide general pre-treatment stand structure and composition. Pre-treatment fuels and vegetation were measured in 1991

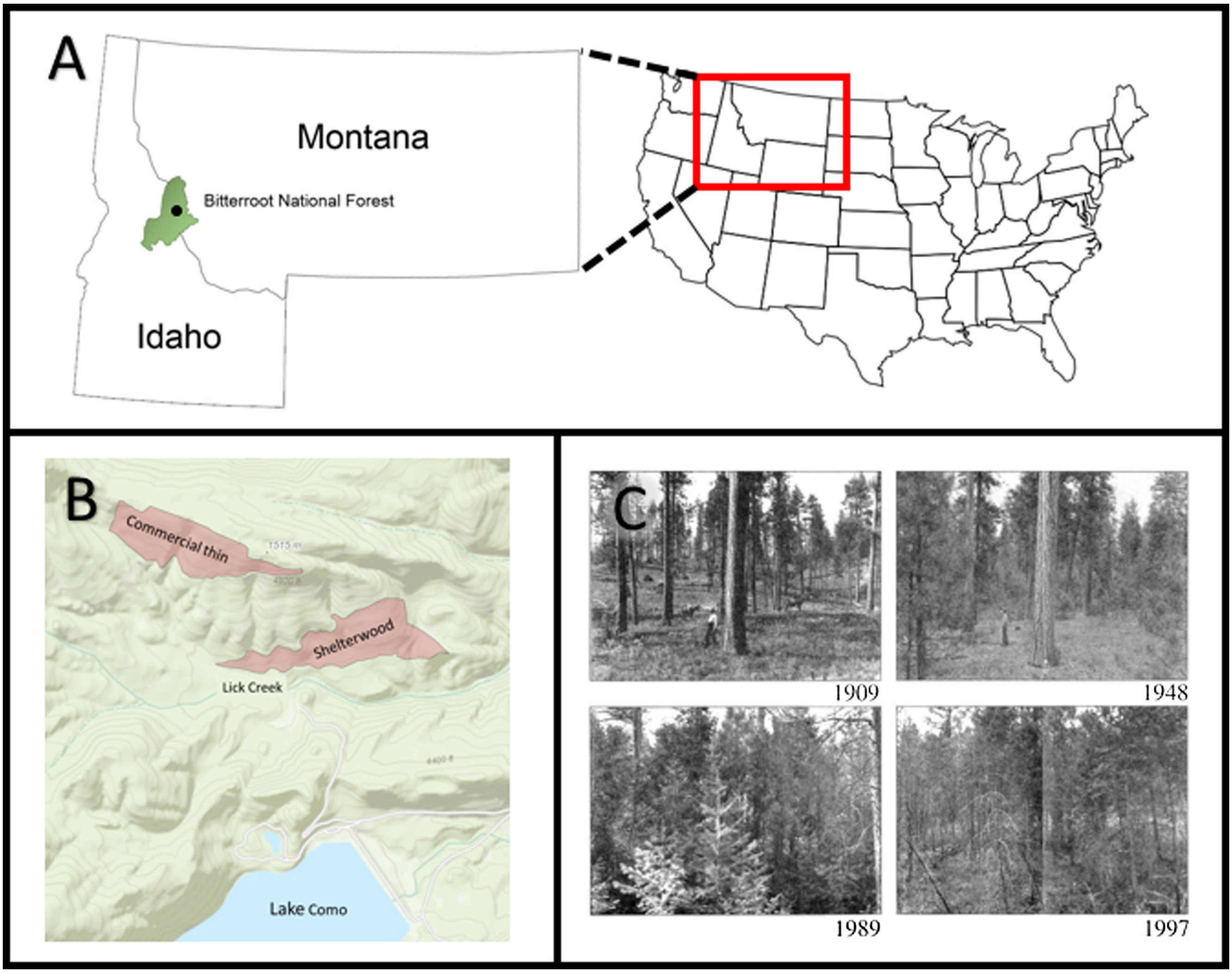

Fig. 1. (a) The study site (black circle) is located on the Bitterroot National Forest in western Montana near the Idaho-Montana state border. (b) The two silvicultural installations are located between 4300 and 5000 ft. in elevation on south facing slopes in the Lick Creek drainage. The shelterwood is located downslope near Lick Creek, while the thinning is upslope, in proximity to the ridge. (c) The Lick Creek drainage is the location of the well-known photo series documenting forest succession after the first large USFS timber sale in ponderosa pine in 1907; photos have been taken approximately every decade since 1909 (Gruell et al., 1982).

Table 1
Summary of the two silvicultural studies: a commercial thinning and a retention shelterwood with various levels of underburning, implemented between 1993 and 1994 (Smith and Arno, 1999). Pre-treatment values in the Control are unavailable as they were not measured prior to treatment.

Treatment	Abbreviation	Pre-treatment		Cutting target	Underburning	Post-treatment	
		TPH	BA	($m\ ha^{-1}$ BA)		TPH	BA
Thinning							
Control	CO	–	–	None	None	454	24
Thin and No Burn	NB	384	21	12	None	220	13
Thin and Spring Burn	SB	435	20	12	1994 Spring burn	310	13
Thin and Fall Burn	FB	447	23	12	1993 Fall burn	279	15
Shelterwood							
Control	CO	–	–	None	None	728	26
Cut and No Burn	NB	534	29	9	None	244	11
Cut and Wet Burn	WB	672	26	9	May 1993 low consumption burn	179	12
Cut and Dry Burn	DB	677	26	9	May 1993 high consumption burn	238	13

in the treated units but not the controls. Harvesting was conducted in July and August of 1992 and prescribed burning occurred between 1993 and 1994. Treatments and early responses were detailed by Smith and Arno (1999).

The thinning is located upslope of the Lick Creek drainage, with a southerly aspect and elevations of 1460 to 1540 m (Fig. 1b). Its silvicultural objective that of a conventional thinning: to maintain the even-aged structure and development of the stand while reducing density and promoting tree growth and vigor. The target residual basal area was 12 $m^2\ ha^{-1}$. The thinning had a pre-treatment average stand age of 70 years, with approximately 369 trees ha^{-1}, 19–23 m^2 of basal area (BA) per hectare, and a 93%

ponderosa pine species composition (Arno, 1999; Harrington, 1999a). The four treatments consist of a control (CO), thinning without prescribed burning (NB), thinning followed by a spring burn (SB), and thinning followed by a fall burn (FB) (Table 1). The 1992 thinning resulted in an average of 219 trees ha^{-1} and BA of 14 m^2 ha^{-1} in the treated units (see Arno (1999) for prescription details). In order to examine the effects of burn seasonality, three units were burned in the fall of 1993 (FB) and three units were burned in the spring of 1994 (SB). See Harrington (1999b) for burning prescription details.

The retention shelterwood is positioned towards the base of the drainage, with a primarily southerly aspect and elevations of 1320 to 1390 m (Fig. 1b). Its silvicultural objective was to initiate the development of a two-cohort stand by recruiting ponderosa pine seedlings and creating conditions for their promotion. The shelterwood cutting aimed to reduce basal area to 9 m^2 ha^{-1}. Prior to the 1992 cutting, the 85 year old stand supported 435 trees ha^{-1}, 27 m^2 ha^{-1} BA, and a 72% ponderosa pine species composition (Arno, 1999; Harrington, 1999a). The four treatments consist of a control (CO), cutting without prescribed burning (NB), cutting followed by a low consumption burn (lower duff was wet; WB), and cutting followed by a high consumption burn (lower duff was dry; DB) (Table 1). The shelterwood cutting resulted in a post-harvest density of 174 trees ha^{-1} and 12 m^2 ha^{-1} basal area in the treated units (see Arno (1999) for prescription details). The WB and DB units were broadcast burned in May 1993. See Harrington (1999b) for burning prescription details. Following the prescribed burns, several dense pockets of regeneration in three of the treated units were precommercially thinned to enhance uniformity of that component.

2.3. Sampling design

All trees and saplings were measured on a systematic grid of 12, 0.04-ha permanent circular plots located within each treatment unit in 1991 (pre-harvest; treated units only), 1993 (post-harvest), 2005, and 2015. Species, diameter at breast height (dbh; 1.4 m above ground), total height, crown base height, crown ratio, crown position, and status (live or dead) were recorded for all trees $\geq$ 10 cm dbh. Species, diameter, crown ratio, and a subset of total height were measured on all saplings ($\geq$1.4 m tall and <10 cm dbh). In 2015, seedlings, forest floor biomass, understory vegetation, and stumps were also measured. We measured seedlings (<1.4 m tall) on a 0.004-ha subplot, nested within the 0.04-ha plot. We recorded species and height class for all seedlings, where heights were categorized by bins centered at 0.06 m, 0.3 m, 0.6 m, 0.9 m, and 1.2 m. Duff, litter, fine woody debris (FWD; $\leq$7.6 cm in diameter) and coarse woody debris (CWD; >7.6 cm in diameter) constitute forest floor biomass, and were measured along two, 16.7 m transects per plot using the planar intercept sampling method (Brown, 1974). The first transect rotated sequentially from upslope to downslope by 45° (i.e. upslope for plots 1,5,9; 45° from upslope for 2,6,10; 90° from upslope for 3,7,11; and 135° for plots 4,8,12). The second transect was located 90° from the first transect. Live and dead shrub and herb understory percent cover and height were estimated in two, 1 m radius plots along each transect. We also recorded stump height, diameter, and decay class on all odd numbered plots within the entire 0.04-ha plot.

2.4. Allometric biomass estimation

We used species-specific allometric equations developed in the interior of British Columbia by Standish et al. (1985) that utilize measured diameter and height to estimate whole-tree aboveground biomass. Per-tree biomass was then summed for each plot and expressed on an area basis. We calculated standing dead tree (snag) biomass using the same Standish et al. (1985) equations, but adjusted for wood decay using species-specific dead:live biomass density ratios developed by Cousins et al. (2015). The decay classes (i.e., 1–5) in Cousins et al. (2015) were designated as either sound (1–3) or rotten (4 and 5). We applied the mean of the sound ratios (0.92 for ponderosa pine, 0.67 for Douglas-fir) to all standing snags with intact tops and the average of the higher decay class ratios (0.58 ponderosa pine; 0.51 Douglas-fir) to all snags with broken tops. Whole-tree biomass equations were used for both live and dead trees with broken tops. While this likely underestimates the biomass in these trees given the taper assumptions associated with whole-tree allometries, there are currently no equations addressing trees with broken tops, and their frequency was very low (just 22 of a total 9380 trees recorded).

For all seedlings, biomass was estimated from height via height-dependent equations developed in western Montana by Brown (1978), who generated whole tree equations for all trees less than 4.6 m tall. Per-seedling biomass was then summed to the plot level and expressed on an area basis.

We estimated aboveground stump biomass using species-specific stump equations generated by Woodall et al. (2010). Volume of stump with and without bark were calculated from top height diameter, stump height, and bark thickness using equations by Raile (1982). As no stump volume estimators currently exist for western conifer species, we used red pine (*Pinus resinosa* Aiton) parameters to estimate both ponderosa pine and Douglas-fir stump volumes. These were then adjusted for differences in wood specific gravity by species (Woodall et al., 2010). Since species was unidentifiable for the majority of stumps, this parameter was assigned by determining the relative proportion of ponderosa pine and Douglas-fir that were harvested (given pre-harvest and post-harvest species compositions), and then randomly allocating species to stumps on that proportional basis. To account for decay, we applied the same Cousins et al. (2015) dead:live ratios used for snags to stumps on the basis of recorded stump decay class (S, R).

We calculated forest floor biomass using planar-intercept sampling methods (Lutes et al., 2006). Understory vegetation biomass was calculated using the surface fuels-vegetation equation available in the FIREMON fire effects monitoring and inventory system, where biomass is calculated as a function of height, percent cover, and bulk density (Caratti, 2006; Lutes et al., 2006). Bulk density was assigned using composite values from multiple sources in FIREMON: 0.8 kg m^{-3} for herbaceous plants and 1.8 kg m^{-3} for shrubs (Caratti, 2006; Lutes, 2016).

2.5. Data analysis

We divided ecosystem biomass components into categories for analysis comparable to similar studies (Boerner et al., 2008; Finkral and Evans, 2008; Campbell et al., 2009; North et al., 2009; Stephens et al., 2009; Sorensen et al., 2011). Tree biomass consisted of all trees $\geq$1.4 m tall, while regeneration biomass included all trees <1.4 m tall. Forest floor biomass elements were analyzed separately by component (duff, litter, FWD, CWD) and pooled. Understory biomass, stump biomass, and snag biomass were analyzed individually. Total aboveground biomass was calculated by summing the biomass of all live and dead components.

All biomass component distributions were severely non-normal, often with heavy right skewedness, and residuals strongly deviated from normality using a preliminary mixed-effects model. Log transformations were not feasible for the forest floor biomass estimates due to zero values, and exponential transformations did not normalize the residuals. Therefore, we used non-parametric permutation testing to analyze differences in biomass levels using several linear contrasts (Manly, 2007). First, we ran either an analysis of variance or simple linear model, with biomass as the response and treatment as the explanatory variable. Biomass observations were then randomly reassigned in each unit across all 12 units 999 times to create a sample of F-statistics (or t-statistics) that represent randomly allocated biomass levels. To test the null model (i.e., biomass is the same across treatments), the F-statistic from the original model was compared to the distribution of simulated F-statistics. If the F-statistic was unusually small (less than the 10th percentile; $p < 0.1$) relative to the simulated F-statistics, then those quantities of biomass are unlikely to have arisen if the null model is true, supporting an alternative hypothesis.

We applied this permutation testing to examine several contrasts for each ecosystem component and the total aboveground biomass. First, an analysis of variance was performed to determine (1) whether biomass in any of the treatments was different than any of the other treatments (CO-NB-SB/WB-FB/DB). Then simple linear modeling was applied to: (2) test whether treating a stand resulted in lower biomass irrespective of treatment type (CO-NB & SB/WB & FB/DB); (3) compare biomass in each of the treatments to the control (CO-NB, CO-SB/WB, CO-FB/DB); (4) contrast each of the burn treatments to the no burn treatment (NB-SB/WB, NB-FB/DB); (5) determine whether biomass differed between the burned treatments versus the no burn treatment (NB-SB/WB & FB/DB); and (6) check for differences in biomass between the spring/wet burns and fall/dry burns (SB/WB-FB/DB). Matched pairs t-testing was used to examine differences in live tree biomass between pre-treatment (1991) and post-treatment (1993, 2005, and 2015) sampling events in each treatment. Normality of sample distributions ($n = 3$) for each treatment was confirmed via a Shapiro-Wilk normality test.

3. Results

The proportional distribution of biomass among components was strikingly similar across all units 23 years after treatment, regardless of installation or treatment. The greatest proportion of aboveground biomass was stored in trees, with little variance regardless of treatment or site: trees constituted 93–95% of the aboveground biomass profile (range incorporates treatment unit averages across all treatments and both installations; $n = 8$). Of the remainder, approximately 2–3% consisted of forest floor biomass: litter, duff, FWD, and CWD. Snags and stumps each accounted for 1–2%, regeneration accounted for 0.01–0.3%, and understory vegetation accounted for just 0.03–0.08%.

3.1. Trees

In both the thinning and shelterwood installations, pre-treatment (1991) tree biomass was the same across each of the fuel reduction treatments (Table 2), illustrating homogeneity of the forest among designated treatment units. Harvesting removed an average of 28.5 Mg ha^{-1} of biomass from the thinned units and an average of 71.9 Mg ha^{-1} from the shelterwood units (Table 2).

In the thinning, tree biomass one year after harvesting (1993) was approximately 67% of pre-treatment biomass, and 57% of biomass in the control. However, there was no difference among fuel reduction treatments, indicating that burning did not immediately further reduce live tree biomass. By 2005, 13 years after treatment, tree biomass across all fuel reduction treatments in the thinning had recovered to pre-treatment levels. By 2015, trees in the fuel treatments stored an average of 127% of pre-treatment biomass (Fig. 2a). In 2015, biomass levels remained similar across fuel treatments, while all three were less than the control by an average of 27% ($p = 0.02$, 0.01, and 0.02 for NB, WB and DB, respectively) (Fig. 3a).

In the shelterwood, average post-treatment (1993) tree biomass in treatments was 47% of the pre-treatment biomass and 40% of the biomass in the control, but also did not differ among fuel reduction treatments (Table 2). In 2005, biomass in the two burned treatments was still significantly lower than pre-treatment biomass levels. In the NB treatment, biomass had recovered by 2005, but it remained lower than the control (Fig. 2b). By 2015, biomass in the NB was 153% of pre-treatment levels, while in the burned treatments, biomass was only 85 and 89% of pre-treatment biomass in the WB and DB units, respectively (Table 2). Burning (WB & DB) resulted in a little more than half of the biomass accumulation of the two unburned treatments (NB & CO), even 23 years after treatment (Fig. 4a).

3.2. Forest floor

In 2015, differences in forest floor biomass at the thinning installation were primarily due to less duff and litter biomass in the fuel reduction treatments versus the control. Thinned stands had between 15 to 23% less litter and 43 to 74% less duff than the control, with the greatest reductions in the burned treatments (Fig. 5). Total forest floor biomass ranged 3 Mg ha^{-1} in the WB to 4.7 Mg ha^{-1} in the control (Table 3). Thinning alone reduced forest floor biomass by 20% relative to the CO, while thinning and burning reduced forest floor biomass by an additional 8 to 16% (FB, SB, respectively) (Fig. 3d). In the shelterwood, burning resulted in lower levels of CWD, FWD, and duff biomass compared to both the CO and NB, reducing total forest floor biomass by 34 to 39% relative to the CO and by 39 to 43% relative to the NB (Fig. 6). Differences in conditions during the burns were apparently inconsequential at both sites, as there was no difference in forest floor biomass between the SB and FB in the thinning or the WB and DB in the shelterwood. In the shelterwood, similarities in forest floor biomass between the WB and DB treatments may be

Table 2
Mean tree biomass (Mg ha^{-1}) pre-treatment (1991), removed from treatment, post-treatment (1993), 13 years after treatment (2005), and 23 years after treatment (2015). * indicates significant differences from pre-treatment (1991) biomass levels at the 0.1-level; ** at the 0.05-level; *** at the 0.01-level.

	1991	Harvested	1993		2005	2015
Thinning						
CO	–	–	102.1		142.0	149.4
NB	88.1	31.9	56.2	***	86.2	110.2
SB	77.5	23.2	54.3	**	78.8	104.7
FB	93.1	30.4	62.8	**	88.4	112.4
Shelterwood						
CO	–	–	156.7		193.6	214.4
NB	146.8	76.8	70.0	**	140.6	224.8
WB	128.9	69.7	59.2	**	72.4	109.7
DB	132.2	69.2	63.0	**	79.2	117.9

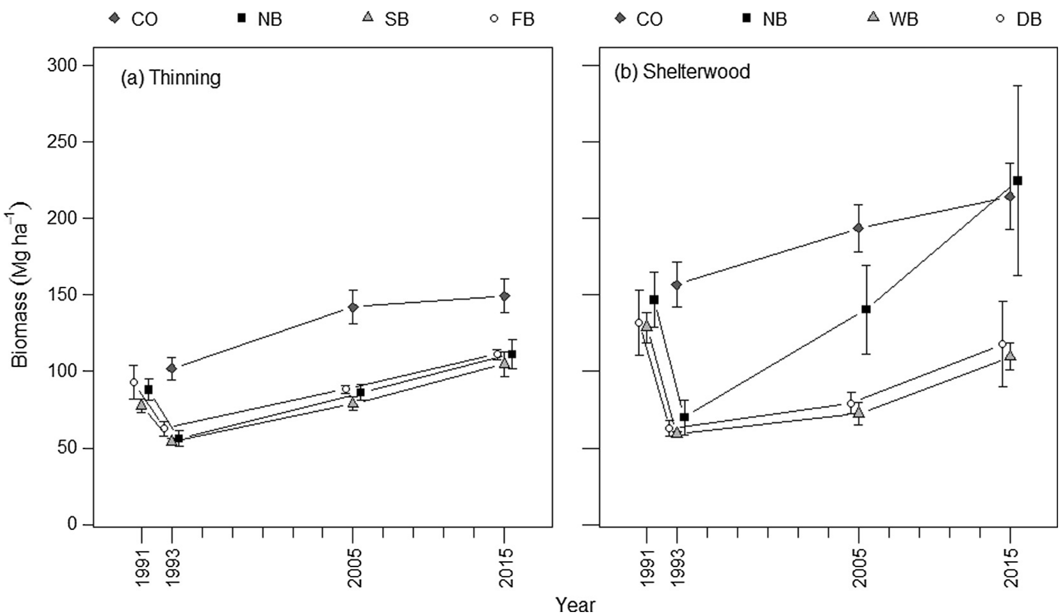

Fig. 2. Tree biomass over time, beginning pre-harvest in 1991. First post-treatment remeasurement (1993) was one year after harvesting. Control data was not collected until 1993.

attributed to a rainstorm that immediately followed burning in the DB treatment, which likely diminished the severity of fire effects in the DB (Harrington, 1999b).

3.3. Stumps and snags

Stump biomass levels were similar across all treatments for both the thinning ($p = 0.44$) and the shelterwood ($p = 0.95$); this included even the controls, due to the ubiquitous presence of stumps from past harvesting in the area (Smith and Arno, 1999) (Figs. 3e, 4e). Snag biomass was homogeneous across all treatments in the thinning ($p = 0.99$) (Fig. 3f). In the shelterwood, however, snag biomass in the control was between 4 and 20 times greater than in each of the fuel treatments ($p = 0.01$ for NB, $p \sim 0.00$ for WB, DB), likely a reflection of competition-induced mortality associated with its higher stand densities (Fig. 4f). Snag biomass was lowest in the burn treatments, and burning further reduced snag biomass relative to the NB treatment by as much as 84% ($p = 0.04$).

3.4. Understory vegetation and seedlings

In 2015, understory vegetation biomass was the same across all treatments in both the thinning ($p = 0.95$) and shelterwood ($p = 0.44$) (Fig. 3c). In the thinning, vegetative biomass ranged 0.05 Mg ha^{-1} (NB and SB) to 0.06 Mg ha^{-1} (CO and FB) (Table 3). Similarly, in the shelterwood, biomass ranged 0.09 Mg ha^{-1} (WB and DB) to 0.14 Mg ha^{-1} (NB) (Fig. 4c). Seedling biomass was also similar across all treatments, including the control, in both the thinning ($p = 0.51$) and shelterwood ($p = 0.96$) (Figs. 3b, 4b). In the thinning, seedlings accounted for 0.02 Mg ha^{-1} (NB) to 0.07 Mg ha^{-1} (FB) (Table 3). In the shelterwood, seedling biomass was much greater, accounting for 0.25 Mg ha^{-1} (CO) to 0.35 Mg ha^{-1} (WB).

3.5. Total aboveground biomass

Total biomass 23 years after treatment at the thinning installation was significantly lower in each of the three fuel reduction treatments (NB, SB, FB) by 25–30% relative to the control's 159.7 Mg ha^{-1} ($p < 0.01$ for all three treatments) (Fig. 7). There was no difference among the three fuel reduction treatments, with the NB units storing an average of 119.9 Mg ha^{-1} of biomass, the SB units an average of 112.2 Mg ha^{-1}, and the FB units an average of 119.2 Mg ha^{-1} (Table 3). At the shelterwood, burning reduced total biomass by 50% in the WB and 45% in the DB relative to the control ($p = 0.08$ for WB and DB each) (Fig. 7). The NB treatment however, resulted in similar levels of aboveground biomass as the control (228 Mg ha^{-1}), with an average of 235.7 Mg ha^{-1}. The WB (116.6 Mg ha^{-1}) and DB (125.2 Mg ha^{-1}) mean biomass levels were a little more than half the biomass in the NB, but only the difference between the WB and NB was significant ($p = 0.05$) (Table 3). There was no difference in biomass between the two types of burning ($p = 0.85$).

4. Discussion

There are several notable long-term responses of biomass components to the fuel reduction treatments. Primarily, our results indicate that tree biomass can return to pre-harvest levels in less than 13 years in some cases, and by 23 years in others, although they remained below 2015 control levels (Fig. 2). Furthermore, at least in the thinning, recovered tree biomass is stored in fewer, larger trees. For example, although biomass levels in 2015 have recovered to 1991 pre-treatment levels in the thinning (Fig. 2), tree densities ranged 29 (NB) to 117 (FB) fewer TPH in 2015 than in 1991 (Clyatt, 2016). In the retention shelterwood, tree biomass in the two burned treatments were just approaching pre-treatment levels, while the NB treatment resulted in biomass that even exceeded the amount before treatment.

The additional tree biomass in the shelterwood NB units can be explained by differentiating between saplings (<10 cm dbh) and overstory trees across the three fuel reduction treatments: while sapling biomass in the burned treatments ranged just 23.5 Mg ha^{-1} (WB) to 30.3 Mg ha^{-1} (DB), the NB treatment had 118.0 Mg ha^{-1} of biomass stored in saplings, an amount representing more than half of the total tree biomass. Those values reflect the development of advanced regeneration that arose in the absence of

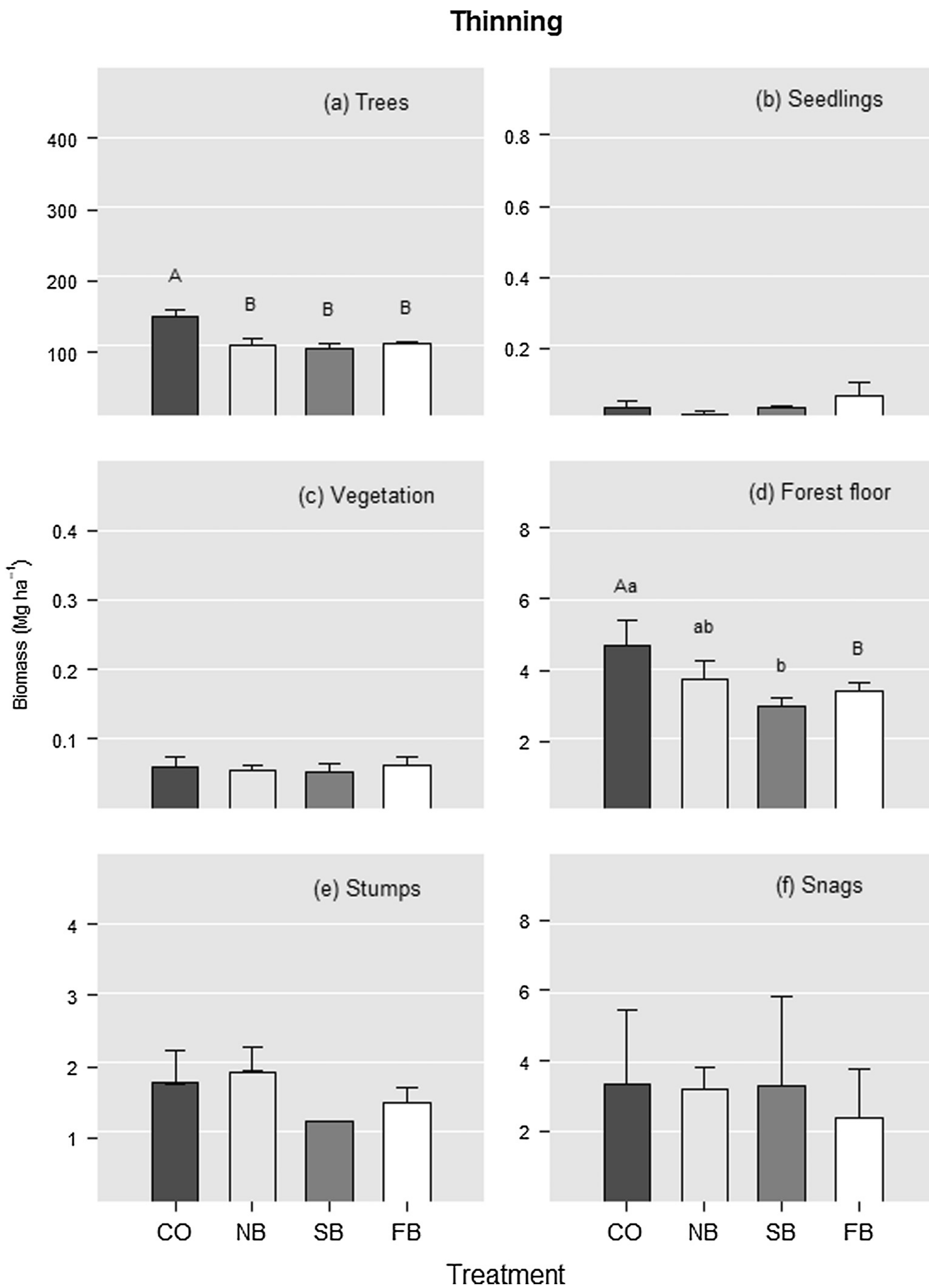

Fig. 3. 2015 average biomass in each of the treatment types (control (CO), no burn (NB), spring burn (SB), fall burn (FB)) for each component in the thinning. Trees (a) include all trees ≥ 1.4 m tall, while seedlings (b) are all trees < 1.4 m tall. Vegetation (c) includes all shrubs and herbs. Forest floor (d) refers to the sum of duff, litter, fine woody debris and coarse woody debris biomass. Stumps (e) are all dead tree remnants < 3 m tall, while snags (f) are dead trees ≥ 3 m tall. Error bars are one standard error from the mean. Letters above whiskers denote significant differences among treatments: uppercase indicates differences at the 0.05-level, while lowercase represents significance at the 0.1-level. Note that biomass scales differ by component.

burning treatments. This result is similar to that from a study examining the effect of various combinations of thinning and burning on rates of biomass accumulation 8 years after treatment in Sierra Nevada mixed-conifer forests. In that study, Hurteau and North (2010) observed that the greatest rate of live-tree carbon accumulation, as well as overall live-tree carbon storage, was accomplished using an understory thinning without burning. In that study, burned units experienced almost half of the carbon gains in large trees over the same duration of time. Such a rapid development of tree biomass was not observed in the NB treatment at the Lick Creek thinning installation, likely because of lower site productivity. The retention shelterwood installation is located in at the base of the drainage, a more productive location as indicated by the presence of mesic species, including grand fir (*Abies grandis* (Dougl. ex D.Don) Lindl.), Engelmann spruce (*Picea engelmannii* Parry ex Engelmann), and subalpine fir (*Abies lasiocarpa* (Hook.) Nutt. var. *lasiocarpa*).

Contrary to our initial hypothesis, it appears that even 23 years after treatment, cutting and burning treatments at both the thinning and shelterwood site continue to maintain lower levels of forest floor biomass than either the untreated or no burn units (Figs. 3d and 4d). In the thinning, lower forest floor biomass was

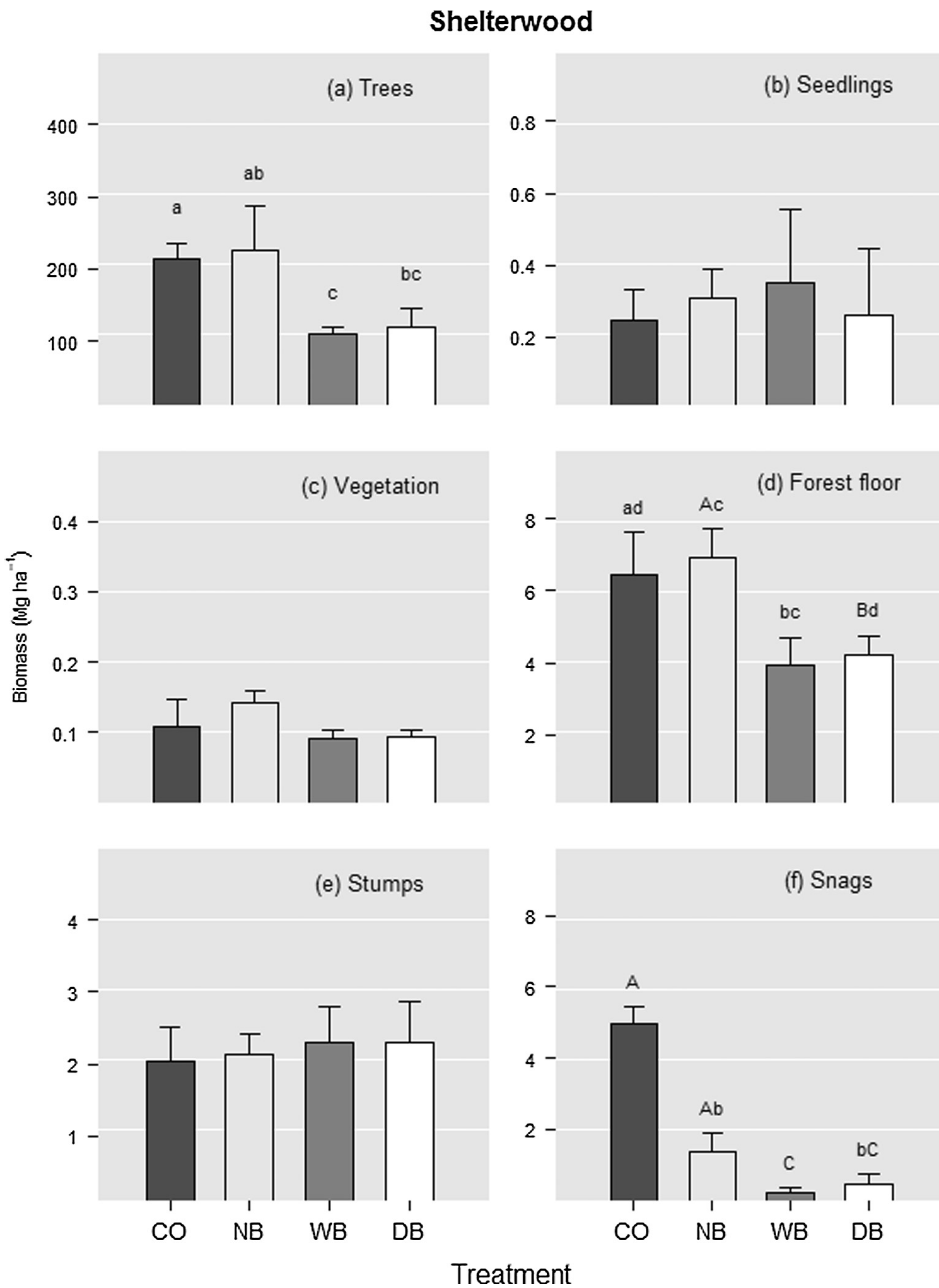

Fig. 4. 2015 average biomass in each of the treatment types (control (CO), no burn (NB), wet burn (WB), dry burn (DB)) for each component in the shelterwood. Trees (a) include all trees ≥ 1.4 m tall, while seedlings (b) are all trees < 1.4 m tall. Vegetation (c) includes all shrubs and herbs. Forest floor (d) refers to the sum of duff, litter, fine woody debris and coarse woody debris biomass. Stumps (e) are all dead tree remnants < 3 m tall, while snags (f) are dead trees ≥ 3 m tall. Error bars are one standard error from the mean. Letters above whiskers denote significant differences among treatments: uppercase indicates differences at the 0.05-level, while lowercase represents significance at the 0.1-level. Note that biomass scales differ by component.

due to less duff and litter in each of the two burned treatments relative to the control (Fig. 4c and d). Litter also tended to be lower in the NB units relative to the control (Fig. 4c). Those results are likely attributed to higher litter decomposition rates in the thinned units, regardless of underburning, due to more sunlight penetrating through to the forest floor. Reduced amounts of duff however, still appear to be an artifact of the initial underburning, even 23 years after treatment. In the shelterwood, only duff was lower for both burned treatments relative to the control (Fig. 5d). CWD also tended to be lower in the shelterwood WB relative to the control, but not the DB (Fig. 5a). The greatest differences in woody biomass in the shelterwood were between the NB and burned units; duff, FWD, and CWD were all lower in the burned units when the WB and DB were pooled. This finding can likely be attributed to the ascension of ingrowth into the overstory, which increased sources of woody debris and likely reduced microbial decomposition activity by decreasing surface exposure to sunlight or precipitation throughfall. Shorter-term studies have produced contradictory results in the response of forest floor biomass to treatments; some detected an initial increase ranging from 18 to 28% more litter and fine woody debris after thinning (Stephens and Moghaddas, 2005a; North et al., 2009), some observed decreases ranging from 10 to 24% (North et al., 2009; Sorensen et al., 2011), and others detected no change at all (Boerner et al., 2008; Finkral and Evans, 2008;

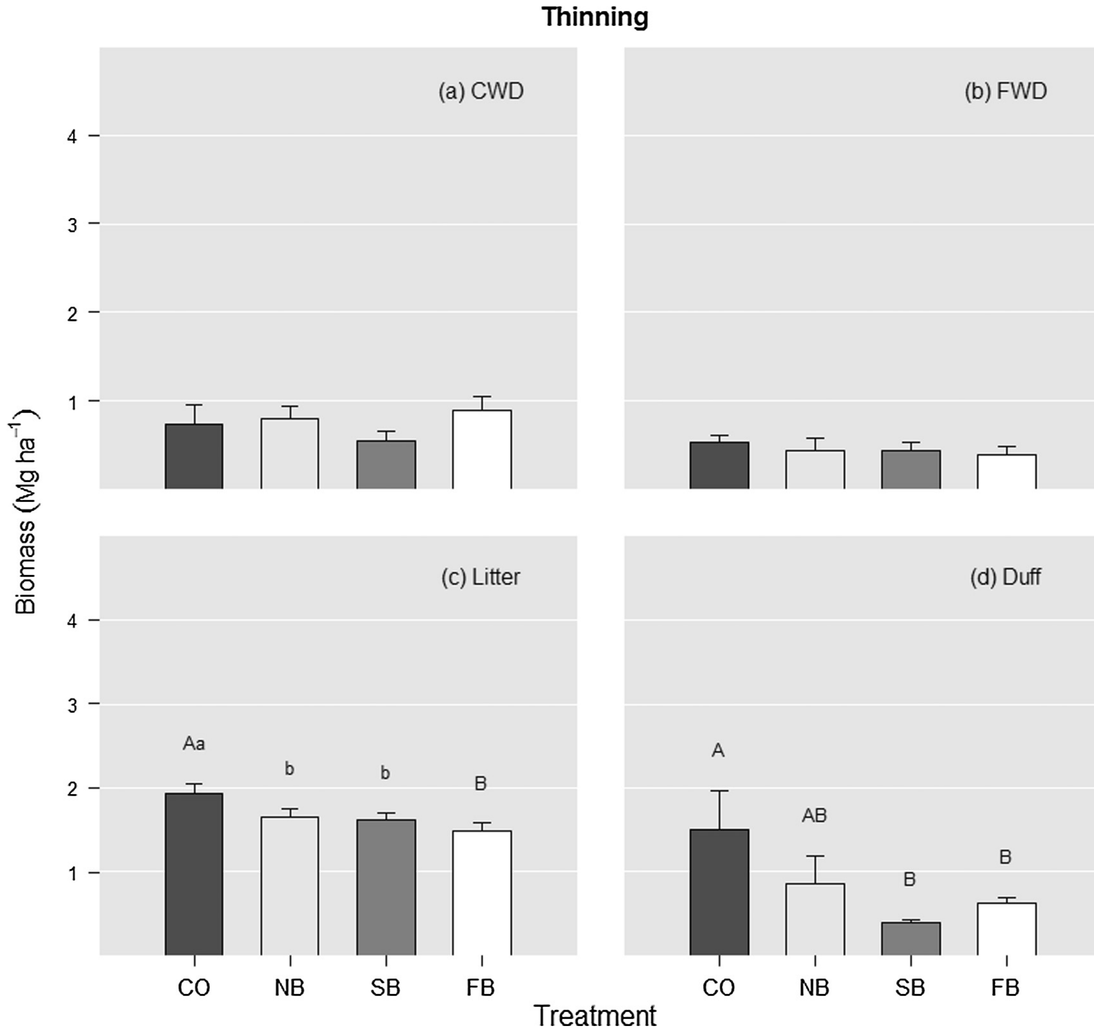

Fig. 5. 2015 forest floor biomass components: coarse woody debris (a), fine woody debris (b), litter (c) and duff (d) in the thinning for each of the four treatment types (control (CO), no burn (NB), spring burn (SB), fall burn (FB)). Alphabetical letters indicate significance at the $\alpha = 0.05$ (uppercase) and $\alpha = 0.1$ (lowercase).

Table 3
Mean biomass (Mg ha^{-1}) 23 years post-treatment for all ecosystem components in each treatment (standard errors in parentheses). Treatments are control (CO), no burn (NB), spring or wet burn (SB/WB) and fall or dry burn (FB/DB). Significant relationships within rows are denoted using alphabetical letters, with capital letters indicating significance at the < 0.05 level and lowercase letters representing significance at the < 0.1 level.

Thinning				
	CO	NB	SB	FB
Trees	149.38 (10.77)A	110.24 (9.51)B	104.66 (7.98)B	112.43 (2.52)B
Seedlings	0.03 (0.02)	0.02 (0.01)	0.03 (0.01)	0.07 (0.04)
Vegetation	0.06 (0.01)	0.05 (0.01)	0.05 (0.01)	0.06 (0.01)
CWD	0.73 (0.22)	0.80 (0.14)	0.55 (0.10)	0.89 (0.16)
FWD	0.53 (0.08)	0.45 (0.14)	0.44 (0.09)	0.39 (0.08)
Litter	1.51 (0.46)Aa	0.87 (0.32)b	0.39 (0.05)b	0.63 (0.07)B
Duff	1.51 (0.46)A	0.87 (0.32)AB	0.39 (0.05)B	0.63 (0.07)B
Stumps	1.75 (0.48)	1.93 (0.33)	1.22 (0.01)	1.51 (0.21)
Snags	3.35 (2.10)	3.23 (0.63)	3.30 (2.53)	2.41 (1.36)
Total	159.27 (8.34)A	119.23 (8.55)B	112.26 (6.21)B	119.88 (2.56)B
Shelterwood				
	CO	NB	WB	DB
Trees	214.44 (21.43)a	224.83 (62.06)ab	109.68 (8.70)c	117.87 (27.59)bc
Seedlings	0.25 (0.09)	0.31 (0.08)	0.35 (0.20)	0.26 (0.19)
Vegetation	0.11 (0.04)	0.14 (0.02)	0.09 (0.01)	0.09 (0.01)
CWD	2.33 (0.68)a	2.58 (0.32)ab	1.12 (0.27)b	1.25 (0.35)ab
FWD	0.84 (0.19)ab	0.91 (0.10)a	0.49 (0.09)b	0.67 (0.12)ab
Litter	1.50 (0.12)	1.64 (0.19)	1.51 (0.43)	1.50 (0.09)
Duff	1.79 (0.22)A	1.80 (0.42)AB	0.81 (0.18)C	0.82 (0.16)BC
Stumps	2.03 (0.46)	2.12 (0.28)	2.29 (0.49)	2.28 (0.57)
Snags	4.99 (0.45)A	1.38 (0.54)B	0.26 (0.12)B	0.48 (0.27)B
Total	228.28 (23.16)a	235.73 (61.79)ab	116.60 (9.36)c	125.23 (28.64)bc

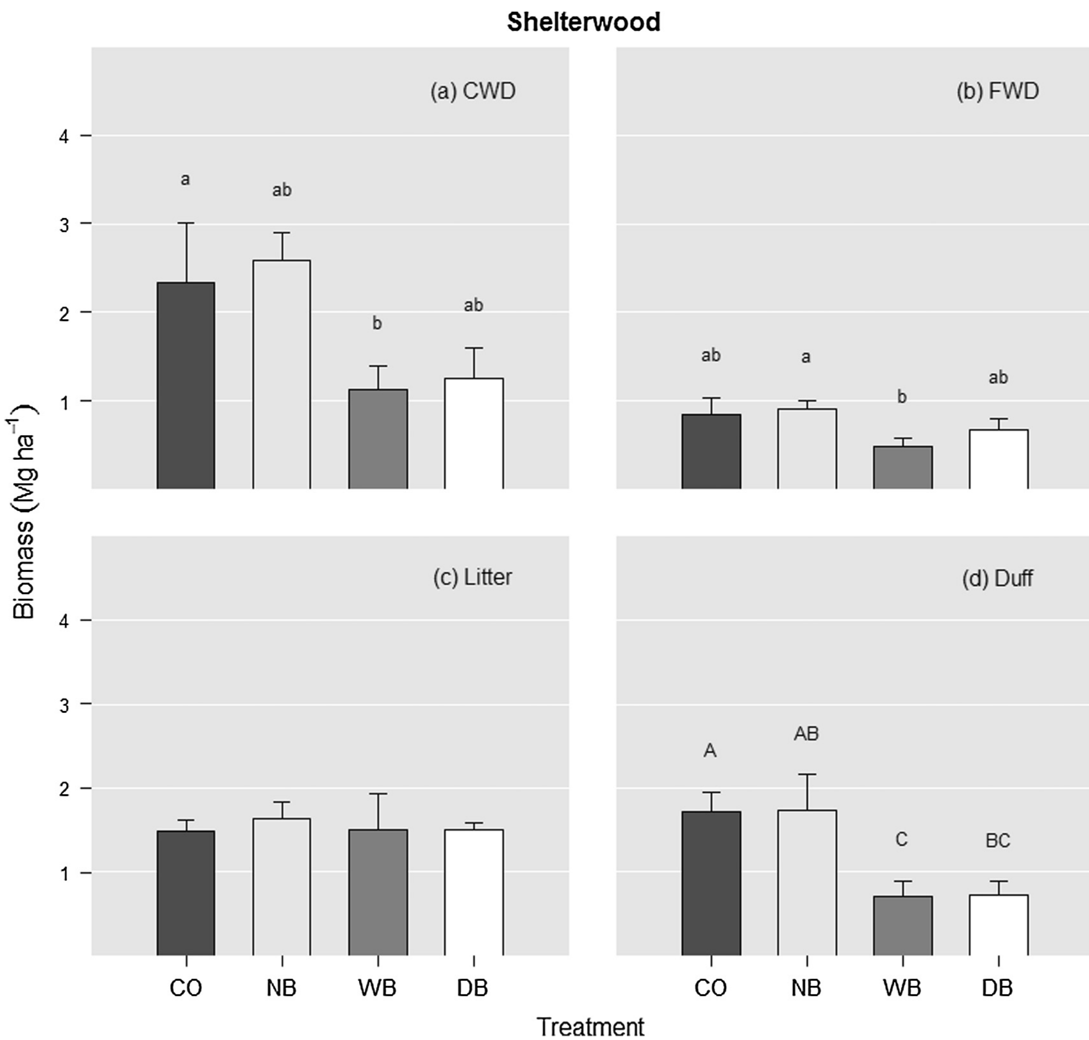

Fig. 6. 2015 forest floor biomass components by coarse woody debris (a), fine woody debris (b), litter (c) and duff (d) in the shelterwood for each of the four treatment types (control (CO), no burn (NB), wet burn (WB), dry burn (DB)). Alphabetical letters indicate significance at the $\alpha = 0.05$ (uppercase) and $\alpha = 0.1$ (lowercase).

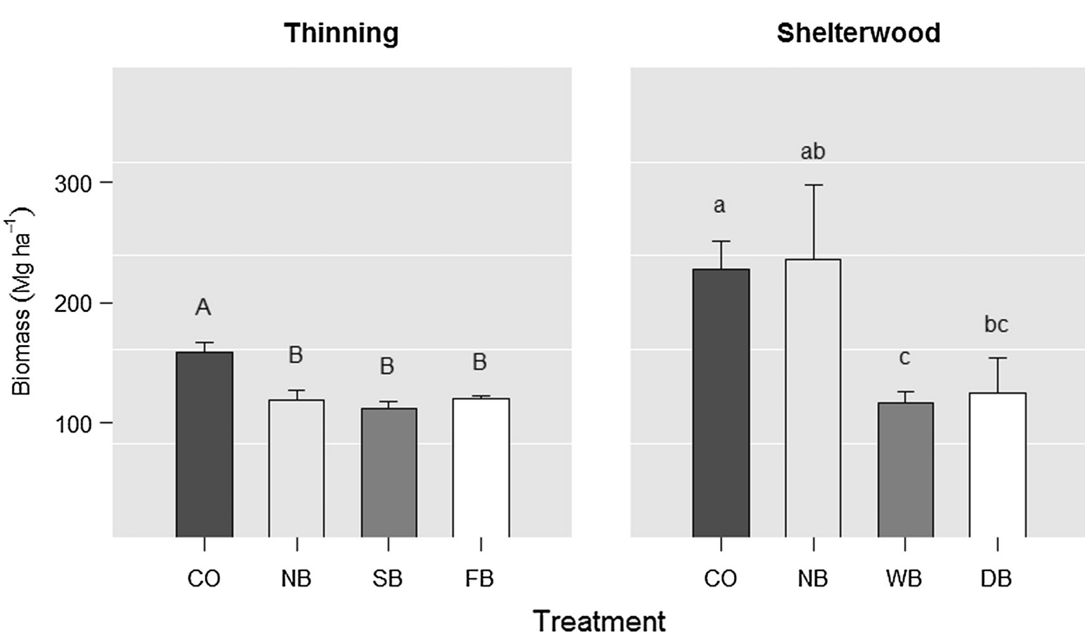

Fig. 7. Total aboveground biomass by treatment for each silvicultural prescription in 2015, 23 after harvesting. Error bars represent one standard error from the mean. Alphabetical letters indicate significance at the $\alpha = 0.05$ (uppercase) and $\alpha = 0.1$ (lowercase).

Stephens et al., 2009). Where broadcast burning was added to the treatment, short-term studies almost unanimously observed a decrease in fine surface biomass (Stephens and Moghaddas, 2005a; Boerner et al., 2008; North et al., 2009; Stephens et al., 2009). This is to be expected, as those fine, relatively dry, woody biomass materials are typically consumed in surface fires.

In our study, snag volumes in treated stands were either the same as controls (thinning) or drastically decreased relative to the control, regardless of underburning (retention shelterwood). Mortality at both installations seemed to be driven by competitive stress and mountain pine beetle (*Dendroctonus ponderosae* Hopkins), with some presence of comandra blister rust (*Cronartium comandrae* Pk.). Other studies have observed an initial increase or no difference in the volume of snags after thinning relative to the control (Stephens and Moghaddas, 2005b; Boerner et al., 2008; North et al., 2009), where mortality can result from logging damage incurred during harvesting. In thinning and burning treatments, snag biomass responses have been mixed, seeming to depending on burn intensity (Stephens and Moghaddas, 2005b; North et al., 2009).

Our study indicates that by 23 years after treatment, understory vegetation is homogeneous across treatment types. In a shorter-term study in a Sierra Nevada mixed-conifer forest, understory biomass increased almost threefold in the thinning treatment relative to the control within the three years following treatment (Campbell et al., 2009). When fuel treatments in fire-frequent forest types involve overstory tree removal, understory production typically increases with improved access to sunlight and below-ground resources (Connell and Smith, 1970; Campbell et al., 2009). However, as the overstory canopy recovers over time, this advantage to understory vegetation diminishes; we would expect vegetation to be similar across treatments in the long term, as was observed in our study.

In fire-dependent forests, such as Lick Creek, managing for carbon storage must be balanced with the need to reduce high-severity fire hazard, as excluding fire is likely not possible in the long-term. Carbon maximizing strategies include lengthening thinning rotations, incorporating mixed species and mixed age stands, and adopting thinning regimes with higher residual stem densities, while strategies to reduce wildfire severity often entails shortening thinning rotations, prioritizing shade-intolerant species, reducing ladder fuels created by mixed-aged structures, and decreasing stem densities (Galik and Jackson, 2009). In the context of fuel reduction efforts, those treatments that explicitly prolong the amount of time required before re-entry can help reduce carbon emissions from harvesting operations or burning, as well as increase carbon sequestration in the stand. Research has consistently shown that thinning and thinning and burning treatments in ponderosa pine forest can restore low-severity fire due to changes in forest structure (Fulé et al., 2012). At Lick Creek, all three fuel reduction treatments in the thinning currently have forest structures apparently conducive to low-severity fires (i.e., higher proportion of ponderosa pine and lower stem density), while in the shelterwood only the two treatments that included broadcast prescribed burning still appear effective (Clyatt, 2016). The high stem density in the shelterwood NB treatment suggests that cutting without broadcast burning at a high productivity site can shorten treatment longevity and will require follow-up treatments in order to maintain the same stand structure as a single cut-and-burn treatment, potentially increasing the carbon cost of maintaining the treatment. Further analyses of potential fire behavior and effects are needed to determine the best options that balance carbon storage with mitigating fuel hazard in fire-dependent forests.

Research has indicated that carbon emissions from treated stands with attributes comparable to the Lick Creek treated stands were significantly lower than untreated stands (e.g. Hurteau and North, 2009; Stephens et al., 2009; Reinhardt and Holsinger, 2010; Yocom Kent et al., 2015). However, it is unclear as to whether total carbon emissions resulting from wildfire occurring in a thinned and burned stand will be lower than those of an untreated stand subjected to wildfire. Future analysis should address this question by modeling carbon emissions from potential wildfires in these and other stands, and determining net carbon loss from treatment-wildfire combinations. Some studies, such as that by Hurteau and North (2010), suggested that carbon emitted from prescribed fires would be sequestered by boosted growth of trees and shrubs within a time period shorter than the historic mean fire return interval due to a carbon fertilization effect. However, others indicate that the probability of a wildfire occurring in a treated stand during the span of treatment longevity is negligible, negating any potential reduction in net emissions (Rhodes and Baker, 2008; Campbell et al., 2012; North et al., 2009). While our research provides information regarding biomass accumulation in the Northern Rockies, future enquiry should incorporate regional disturbance regimes into analysis of potential emissions tradeoffs.

5. Conclusions

The Lick Creek studies analyzed here provide a unique opportunity to examine long-term biomass dynamics and compare treatment responses across two sites with different treatment histories, site productivities, and topographic locations, while holding climate and geographic variables constant. Our results indicate that stands can recover harvested biomass from fuel reduction treatments in as little as 13 years while apparently still achieving restoration objectives, especially if harvesting is followed by prescribed burning. Furthermore, burning resulted in lower levels of forest floor biomass even 20 years after treatment, while results in the no burn treatments were highly dependent on site productivity. Combined with analysis of temporal changes in fuel loads and fire hazard in the decades following treatment, this study's empirical data provide a basis for subsequently determining the relationship of carbon storage and wildfire hazard mitigation in fire-frequent forests of the northern Rocky Mountains.

Acknowledgements

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