# **FINAL REPORT**

Title: Lick Creek Demonstration-Research Forest: 25-Year Fire and Cutting Effects on Vegetation & Fuels

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## **Keywords**

*Pinus ponderosa*; ponderosa pine, fire-prone forest; fire surrogate; hazard fuel treatment; thinning; retention shelterwood; broadcast burning; prescribed fire; resilience; disturbance; fuel treatment longevity; carbon storage; drought; stable-carbon isotopes; exotic species; biodiversity; dendroecology; forest stand dynamics; silviculture; ecological restoration.

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#### Abstract

Fuels reduction treatments are common in ponderosa pine ecosystems of the interior western United States, but the long-term effects on many key ecosystem attributes remain poorly understood, including: tree growth and mortality; forest fuel loads; understory vegetation diversity and composition; production and distribution of aboveground biomass; and physiological response of trees to drought stress. A 1992 experiment at the Lick Creek Demonstration / Research Forest in western Montana was analyzed to evaluate tradeoffs among alternative cutting and burning strategies in ponderosa pine – Douglas-fir stands. One portion of the experiment tested a commercial thinning strategy, while a second tested a retention shelterwood strategy. Units were burned one-to-two years after harvesting, using different broadcast prescribed fire treatments to simulate a range of burning conditions.

All treatments led to a growth release that persisted to the time of re-sampling (23 years; 2015). Changes in stable-carbon isotope relationships were pronounced. Reduction in competition enabled trees to fix carbon and incorporate it into new stem growth when the climate became sufficiently stressful to drive slower-growing trees in uncut stands to either cease new assimilation or become more dependent on stored carbohydrates. During the second posttreatment plot census interval (2005–2010), when mountain pine beetle activity increased locally, tree mortality rates under each control more than doubled compared to the respective treatments. Treatments maintained substantially lower canopy fuel loads (and lower canopy bulk density in thinning units), and tended to lower 1-hr fuels, litter, and duff. The heavier cutting associated with the shelterwood accelerated ladder fuel development, reducing canopy base height. Understory cover (except forbs) was initially reduced by treatments, but then returned to pretreatment levels within 15 years. Species richness increased, and then declined, but remained slightly higher than pretreatment. Forbs responded most strongly to treatments. Understory cover was negatively related to overstory basal area (but species richness and composition were unrelated to basal area). Across all treatments, tree biomass recovered to pre-harvest levels by 2015 (after 23 years). In the thinning, the control exhibited greatest total aboveground and livetree biomass, but those did not differ among the three cut fuel 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 tended to be lower in the burned treatments. Seedling, vegetation, and stump biomass were similar across treatments.

Fuel treatment longevity was strongly influenced by the initial silvicultural prescription, which produced divergent fuel loads and fuel structures. Stand density reduction was key to improving the ability of residual trees to tolerate climatic stress and associated biotic disturbances. Faster growth and enhanced ability to assimilate carbon under more stressful climate following treatments became evident as tree survivorship increased. Understory vegetation was resilient to the treatments in the long term. However, deviation in species composition and non-native invasion occurred by treatment, indicating that the more severe the treatment the greater the deviation from pre-treatment and greater non-native understory vegetation. Treated stands recovered tree biomass to pre-harvest levels in less than 23 years, while yet exhibiting stand densities and fuel loads that foster resilience and advance forest restoration objectives.

# **Objectives**

This project was proposed in response to the Task Statement 7 (of Project Announcement No. FA-FON0015-0001) titled, "Re-measurement – long-term fire effects on vegetation and fuels." Proposals were sought, "to re-measure existing long-term (15 or more years post-fire) field studies of wildfire or prescribed fire effects on vegetation and fuels. The rationale behind the task was that, "A better understanding of long-term vegetation and fuels succession is needed to integrate management objectives for fire into ecosystem restoration and hazardous fuels projects, evaluate changes to ecosystem services, and to assess possible impacts related to climate change."

Our study very closely addressed this task and its rationale. We proposed revisiting and an experiment consisting of a variety of ecosystem restoration and hazardous fuel treatments that included variants of burning treatments in combination with cutting treatments. Our analyses would report on changes and trends among a variety of metrics more than 20 years following treatment. In so doing, we also intended to extend the lifespan of this important study area by refurbishing the study's field monumentation; by rebuilding key research relationships among RMRS, university, and Bitterroot NF staff; and by updating, synthesizing, and improving accessibility of the study's historic data and metadata.

We were able to achieve all of the objectives that we had proposed to accomplish with this project, as well as some additional objectives that we incorporated during the course of the study. Our primary goal was to apply the Lick Creek study and its relatively long response period to address key questions and that could inform effective forest restoration management. We intended to address those questions using data from the study's seven experimental restoration treatments: control; retention shelterwood cutting; retention shelterwood + wet prescribed burn, retention shelterwood + dry prescribed burn; commercial thinning; thinning + Fall prescribed burn; and thinning + Spring prescribed burn. We had three overarching study questions:

- 1) How have restoration cutting and burning treatments affected fuel loading?
- 2) How have restoration cutting and burning treatments affected understory vegetation?
- 3) How have restoration cutting and burning treatments affected ponderosa pine forest resilience to: a) drought; b) mountain pine beetle; and c) fire hazard.

Through efficient cooperation with another funding source, during the course of the study we were able to add the following question:

4) How have restoration cutting and burning treatments affected biomass recovery?

We hypothesized that tree density had increased in all treatments, and that the increase was largely due to Douglas-fir ingrowth. Treatments with cutting and burning would also have increased ponderosa pine seedling and sapling establishment and faster annual growth compared to the control treatments. We expected, however, that treatment benefit on residual tree growth would have decreased relative to 2001 (Sala et al. 2005), due to increased density and canopy cover since then. We also predicted cut + burn treatments would have higher tree growth and

treatment longevity than the cut-only treatments, because the burns killed many seedlings and saplings. We anticipated that surface and ground fuels would have increased in all treatments over time, but we predicted loading to be highest in the control, followed by the cutting treatments, with the treatments including burning to have the lowest loading. We predicted that treatment differences in vegetation and fuel dynamics would translate to differences in forest resilience from drought, wildfire, and bark beetles.

Beyond those research questions, we had a secondary goal of utilizing this research project experience as a vehicle for restoring structural integrity and long-term viability to this high-quality research asset. For this secondary goal, our objectives were to:

- Collate all existing Lick Creek study data, and compile them with new data collected for this project in one archived, accessible Forest Service dataset
- Update and improve all research installation monumentation
- Update the historic Lick Creek photopoint series with a new round of photos, and compile all photographs in one archived, accessible Forest Service dataset
- Provide a solid foundation for future protection and stewardship of the Lick Creek site, by restoring professional working relationships between the Rocky Mountain Research Station, the Bitterroot National Forest, and the University of Montana

# Background

Knowledge of forest vegetation and fuel dynamics following restoration treatments, and how these differ among restoration treatment alternatives, is essential for managers to understand and prescribe treatments with efficacy and longevity. In the northern Rocky Mountains, firedependent ponderosa pine forests were historically maintained by frequent, low severity fires. Reduced wildfire occurrence since the early 1900's has led to denser forests with increased surface and ladder fuels in many areas. Managers often use a variety of cutting, burning, and cutting + burning treatment combinations to achieve ecosystem restoration and hazardous fuels reduction objectives (hereafter: "restoration treatments") in areas with altered fire regimes. Research has demonstrated the short-term success of many treatments to restore forest vegetation structure and composition to a more desirable ecological state and to minimize the occurrence of uncharacteristically high intensity, stand-replacing fires. However, the long-term effects of those restoration treatments on vegetation, fuel dynamics, resilience, and biomass recovery remain unclear. As a result, managers of ponderosa pine forests in the Northern Rockies lack proper guidelines to anticipate the longevity of alternative restoration and fuel treatments, to assess the need to maintain such treatments, or to determine the frequency at which maintenance should occur (Jain et al. 2012).

The Lick Creek Demonstration/Research Forest (Darby Ranger District of the Bitterroot National Forest) offered us a truly unique opportunity to assess 25-year-effects of cutting and burning restoration treatments. Many managers would readily recognize Lick Creek as the site from which iconic images documenting forest change during the fire exclusion era were developed from a photographic series dating from 1909 to 1997 (Smith and Arno 1999). On a hillslope in

that same in the same Lick Creek watershed, a cooperative venture among the Bitterroot National Forest, University of Montana, and Forest Service Intermountain Research Station (now Rocky Mountain Research Station) in 1991 initiated a new manipulative research experiment to explore a variety of treatment strategies to restore the site's ponderosa pine vegetation community and reduce fuel loads down to historically-appropriate levels (Smith and Arno 1999). Two separate but closely related experiments totaling 215 ha were designed, with cutting and prescribed burning treatment variants that sought to restore the site through varying strategies. In doing so, they embraced virtually the full suite of possible treatment combinations that are currently employed by managers of ponderosa pine forests in this region. Silvicultural treatments were implemented in 1992, followed by prescribed burning in 1993 and 1994, under a fully replicated experimental design involving randomization of treated units and a permanent, systematic plot sampling network. In a formal recognition of its long-term research value, the site was officially designated as a Demonstration/Research Forest by the Bitterroot National Forest to encourage its integrity as a long-term research site.

The Lick Creek Demonstration project offered us an unparalleled opportunity to gain understanding of more than 25-year responses of vegetation and fuels to ponderosa pine restoration treatments (1991-2018). No other study of this duration exists in the Northern Rockies. Moreover, the inferential value of treatments employed at Lick Creek is very high: the forest type is ubiquitous in the northern Rocky Mountains, and the treatments performed more than 25 years ago have become staples of ponderosa pine forest management in this region. Additionally, treatment implementation was meticulously documented, the sampling plot network was fully intact, the stands remained unmolested, and historic data records were complete available.

#### **Materials and Methods**

#### **Study Area**

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) (Figure 1). 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 meters, 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).



**Figure 1.** The study site (a) is located on the Bitterroot National Forest in western Montana near the Idaho-Montana state border. The two treatment installations (b) are located between 4300 and 5000 ft. in elevation on south facing slopes in the Lick Creek drainage. The retention shelterwood is located downslope near Lick Creek, while the thinning is upslope, in proximity to the ridge. The Lick Creek drainage is the site of a well-known photo series (c) documenting forest succession and management 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).

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).

Similar to other ponderosa pine/Douglas-fir forests in the northern Rockies (Heyerdahl et al. 2008), the historic, pre-settlement fire return interval across the Lick Creek drainage averaged seven years (ranging five to fifteen years) (Gruell et al. 1982) and was characterized by low-intensity surface fires (Arno 1976, Arno and Fiedler 2005). 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).

#### **Experimental Design**

Two installations were examined: a commercial thinning and a retention shelterwood that were concurrently established as independent studies, each with a complete block design and subsampling (Figure 2). 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 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 meters. 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 (Table 1; Figure 3). The target residual basal area was 12 m<sup>2</sup> ha–1. The thinning had a pre-treatment average stand age of 70 years, with approximately 369 trees ha–1, 19-23 m<sup>2</sup> of basal area (BA) per hectare, and a 93% ponderosa pine species composition (Arno 1999a, 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 m2 ha–1 in the treated units (see Arno (1999a) 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.



**Figure 2.** Thinning (top) and Retention Shelterwood (bottom) with treatment units color coded as follows: "CO" for control, "NB" for cut but unburned units, "WB/SB" for Cut and Wet Burn (Retention Shelterwood) / Cut and Spring Burn (Thinning), and "DB/FB" for Cut and Dry Burn (Retention Shelterwood) / Cut and Dry Fall (Thinning).

The retention shelterwood is positioned towards the base of the drainage, with a primarily southerly aspect and elevations of 1320 to 1390 meters. Its silvicultural objective was to initiate the development of a two-cohort stand by recruiting ponderosa pine seedlings and creating conditions for their promotion (Table 1; Figure 3). 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 m2 ha–1 BA, and a 72% ponderosa pine species composition (Arno 1999a, 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 m2 ha–1 basal area in the treated units (see Arno (1999a) 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.

**Table 1.**Summary of pre-treatment and post-treatment forest conditions. Each treatment<br/>consisted of a cutting (1992) followed by two types of underburning (plus no<br/>burning) conducted in 1993-94. Pre-treatment values in Control were not measured.

#### Thinning

Treatment	Pre-treatment		Cut target	Underburning	urning Post-treatment	
	TPH	BA	(m ha <sup>-1</sup> BA)		TPH	BA
Control (CO)	-	-	-	None	454	24
Thin & No Burn (NB)	384	21	12	None	220	13
Thin & Spring Burn (SB)	435	20	12	1994 Spring burn	310	13
Thin & Fall Burn (FB)	447	23	12	1993 Fall burn	279	15

#### **Retention Shelterwood**

Treatment	Pre-treatment		Cut target	Underburning	Post-treatment	
	TPH	BA	(m ha <sup>-1</sup> BA)		TPH	BA
Control (CO)	-	-	-	None	728	26
Cut & No Burn (NB)	534	29	9	None	244	11
Cut & Wet Burn (WB)	672	26	9	May 1993	179	12
Cut & Dry Burn (DB)	677	26	9	May 1993	238	13

#### Thinning

#### **Retention Shelterwood**



**Figure 3.** Time series of representative plots, with photos taken at the pre-treatment phase in 1991 (top row) and at the time of this study in 2015-16 (bottom row). Examples of the Cut and Spring Burn treatment in the Thinning (left), and the Cut and Wet Burn treatment in the Retention Shelterwood (right).

#### **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 diameter) and coarse woody debris (CWD; > 7.6 cm 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.

#### Biomass Recovery

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 9,380 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).

## Fuel Loading

Trees and saplings were measured in 0.04-ha circular plots centered at each plot center. All trees  $\geq$ 15.24 cm DBH were measured in 1991 prior to harvest (treated units only), in 1993-4 after prescribed burning, and again in 2005 and 2015. Each tree's species, diameter, height, crown base height (post-harvest only), and condition (live/healthy, unhealthy, dead) were recorded. Saplings ( $\geq$ 2.50 cm and <15.24 cm in the thinning and >0.10 and <15.24 cm in the shelterwood) were measured in 1991, and saplings >0.10 and <15.24 cm were measured in all following visits in both installations (1993-4, 2005, and 2015). Sapling species and diameter were recorded. A subsample of systematically selected saplings, for a minimum of 10% sampling, was measured for height, crown base height, and crown ratio. Seedlings were tallied by species in 0.004-ha nested circular subplots centered on each plot center. We summarized overstory structure metrics – stem density, quadratic mean diameter, and basal area – with mean and standard error by installation and treatment.

In cut-burn treatment units (thinning SB/FB and shelterwood WB/DB) following cutting, woody surface fuels were quantified via one Brown's (1974) planar intersect transect per plot. These transects were permanently monumented with metal duff spikes at the start (plot center) and end points (15.24 m). Fuels were measured prior to the burn treatments (spring 1993 in both installations) and again following burn treatments (spring/summer 1993 in the shelterwood, fall 1993 in the thinning FB and spring 1994 in the thinning SB). Woody surface fuels were distinguished by size-based time-lag diameter classes of 1-hr (<0.64 cm), 10-hr ( $\geq$ 0.64 and <2.54 cm), 100-hr ( $\geq$ 2.54 and <7.62 cm) fuels, and sound or rotten 1000-hr fuels ( $\geq$ 7.62 cm). Along each transect, 1-hr fuels were measured from 0-0.30 m, 10-hr fuels were measured from 0-1.80 m, 100-hr fuels were measured from 0-3.70 m, and 1000-hr fuels were measured from 0-15.24 m. In 2005, transects were remeasured, and transects expanded to all units, with two additional live and dead surface fuels measured at two points (4.60 m and 9.10 m) on each transect: (1) litter and duff depth and (2) live/dead herb and shrub height and percent canopy cover. In 2015, all transects were remeasured, with the addition of overall average fuel bed depth (m) taken at two points (4.60 m and 9.10 m) on each transect.

Surface fuel loadings (kg ha-1) were calculated from the transect data using the FEAT/Firemon Integrated (FFI) software program (Lutes et al. 2006), which utilizes Brown's (1974) and Brown et al.'s (1982) formulas. Surface fuels were categorized in two groups by particle type: fine woody debris (FWD) consisted of 1-, 10-, and 100-hr fuels, and coarse woody debris (CWD) consisted of sound and rotten 1000-hr fuels.

To analyze canopy fuels, we used FuelCalc (Lutes et al. 2016), a software program designed to compute surface and canopy fuel loading at the plot level from measured tree data (species, diameter, height, crown ratio and/or crown base height, and crown class), sapling composition, surface fuel loading, and understory vegetation cover. Because we had a subset of sapling data

from 2015, we established height-diameter equations for each predominant species (PIPO and PSME) by installation. These equations were then used to generate fitted heights for all remaining sapling records. All saplings were categorized as "intermediate" crown class and assigned 50% crown ratio.

Overstory tree characteristics, saplings, surface fuels, and vegetation input data from FFI were used by FuelCalc to calculate canopy fuel loading, canopy bulk density, and canopy base height. Live fuel loadings (herbaceous biomass and shrub biomass) were derived from FFI, which calculated biomass from the measured heights and percent cover taken along each Brown's transect. Plot-level canopy base height (CBH) and canopy bulk density (CBD) were calculated from stand data by FuelCalc and summarized to the unit level for our inputs. FuelCalc defines CBH as the lowest height above ground where CBD reaches a threshold value: the maximum stand-level CBD x 0.1 up to 0.12 kg m-3, after which 0.012 kg m-3 is used. Canopy bulk density, the mass of canopy fuel loading per unit volume (Scott and Reinhardt 2001), is estimated at the plot-level as the maximum 1.52-m running average in the fuel profile (Lutes et al. 2016).

#### Understory Vegetation

Understory vegetation in treatment units was measured six times: one measurement per year in 1991 (pre-treatment), 1993 (+0 years since treatment, i.e. 1 year postharvest and immediately postfire), 1995 (+3 years), 1997 (+5 years), 2007 (+15 years), and 2015 (+23 years). Control units were only measured in 2007 and 2015. Twelve permanent sampling points were systemically installed in each treatment unit. Four 1 m2 (0.7 m  $\times$  1.43 m) permanent understory vegetation plots (sub-plots) were established per sampling point (plot). The sub-plots were located 2.1 m from the sampling point along and perpendicular to slope contour, and oriented with long axis perpendicular to slope. All understory vegetation species (including small trees, shrubs, forbs, and graminoids) were identified, and cover classes were visually estimated following Daubenmire's (1959) protocol, except for the 2015 measurement, which was conducted using FIREMON inventory protocol (Caratti, 2006). The Daubenmire and FIREMON inventory protocols included seven (0-5%, 5-20%, 20-40%, 40-60%, 60-80%, 80-95%, and 95-100%) and twelve cover classes (0-1%, 1-5%, 5-15%, 15-25%, 25-35%, 35-45%, 45-55%, 55-65%, 65-75%, 75-85%, 85-95%, and 95-100%), respectively. We used cover class midpoints for quantitative analyses (Gendreau-Berthiaume et al., 2015). The sub-plot cover and species richness were summarized in the plot level. In addition, overstory attributes were measured in 1993 (+0 year) and 2015 (+23 years). Diameter at breast height was recorded for all trees taller than 137 cm on 0.04 ha circular plots (11.3 m radius) that were centered over the permanent sampling points. Nomenclature and origin (i.e., native vs. non-native) determination followed the USDA PLANTS Database (USDA NRCS, 2017) and Mincemoyer (2013).

We compared understory percent cover and species richness measurements by treatment. The plant cover and species richness data were grouped by life form (shrub, forb, and graminoid) and by origin (i.e., native vs. non-native). Thus, six models (3 life forms; 2 origins) per response variable (cover and species richness) were constructed. We fitted linear mixed-effects models to account for the temporally correlated error structure generated from repeated measures, treating the experimental unit as a random effect.

Cover data and species richness (i.e., count) were assessed using a generalized linear mixedeffects model with a lognormal and negative binomial distribution, respectively. Treatment, measurement year, and their interaction were tested as explanatory variables. When the explanatory variables were statistically significant ( $\alpha$ =0.05), we used specified linear contrasts to test the comparative treatment effects. The linear contrasts were set to compare the difference in understory vegetation responses among treatments ( $\alpha$ =0.10 with adjusted p-values). The linear contrasts were tested simultaneously, p-values were adjusted by the simulation method (Hsu and Nelson, 1998). All models and contrasts were fit using PROC GLIMMIX in the statistical software SAS 9.4.

We also calculated overstory basal area and included it as a covariate in the above models to detect the effects of retained overstory trees on understory vegetation. Overstory basal area at +23 years was significantly correlated with +0 years and basal area increment during 1993-2015, suggesting multicollinearity may affect model interpretation. Consequently, only the +23 years understory vegetation models included overstory basal area as a fixed effect.

Constrained correspondence analysis (also known as canonical correspondence analysis; CCA) (ter Braak, 1987) was used to investigate the effects of treatment, measurement year, and their interaction on vegetation community. This analysis is essentially a hybrid of an ordination method (correspondence analysis) and regression analysis, providing a useful way to test the effects of explanatory variables on biological communities (ter Braak and Verdonschot, 1995). The basic principle of CCA is to identify the linear combination of constraints (e.g., environmental variables or treatments in this study) associated with the maximum dispersion of species scores (ter Braak, 1987). We then used a permutation test to test differences in understory vegetation composition by the specified constraints. The permutation test compared the observed constraints' inertia (weighted variance) with randomly permuted and refitted constraints' inertia across 1000 iterations (Oksanen et al., 2017). The CCA and permutation test were conducted with the vegan package (Oksanen et al., 2017) in R (R Core Team, 2018).

We used an indicator species analysis to identify key species responses to treatments (De Cáceres et al., 2010; De Cáceres and Legendre, 2009). Indicator species analysis uses an indicator value index, which is maximized when a species is found exclusively or abundantly in a specific treatment (Dufrêne and Legendre, 1997). We conducted this analysis using the explanatory terms (treatment × measurement year) that were identified as statistically significant ( $\alpha$ =0.05) in the CCA permutation tests, while frequency values were used as the response (Livingston et al., 2016). Data were also pooled by treatment type to test the effects of time, and then pooled by time to test the effects of treatment on indicator species. This analysis was run with 1000 permutations using the indicspecies package (De Cáceres and Jansen, 2016) in R (R Core Team, 2018). We used understory vegetation species with mean relative cover (proportion to total plot cover) greater than 5% for CCA and indicator species analyses.

#### Resilience

In summer, 2016, we collected increment cores were collected from 384 ponderosa pine trees. We cored 16 trees per unit in each experiment, providing 48 trees per treatment and 192 trees per experiment. The subset of cored trees was determined by randomly selecting 8 of the 12 plots within each unit and coring the nearest large (> 25.4 cm dbh) and small (< 25.4 cm dbh) tree to the plot center. We collected two cores per tree, on opposite sides of the tree at a mean height of 51 cm, with all cores 5.15 mm in diameter.

The first core per tree was used to calculate growth metrics: basal area increment (BAI), earlywood area increment (EWAI), latewood area increment (LWAI), and the proportion of latewood (PLW, where PLW = LWAI/BAI). We used the second core to calculate BAI before selecting a subset of these cores for stable-carbon isotope analyses (below). We scanned the cores at 1,200–2,400 dpi and measured ring width (all cores) and EW and LW width (the first core per tree) to the nearest 0.001 mm using CDendro Version 9.2 (Cybis Elektronik & Data AB, 2018a). Crossdating was validated using COFECHA (Holmes, 1983).

To calculate BAI, EWAI, and LWAI, we first estimated the distance to the pith from the first ring in each core using CooRecorder Version 9.2 (Cybis Elektronik & Data AB, 2018b). We then summed this distance plus the radial increment for each ring to produce the bole radius (inside bark) at the end of each year of growth (rt). Then we calculated BAI as  $\pi(r_t^2-r_t-1)^2$ ). We followed the same procedure to calculate EWAI and LWAI, with the exception that the outer radius for EWAI was the sum of all preceding ring widths plus the current EW. This value was then used as the inner radius for calculating LWAI. We averaged BAI across the two cores per tree to evaluate trends in BAI over time. For PLW, we used BAI values from only the core on which EW and LW were measured.

A subset of 72 cores from the shelterwood experiment (37.5% of the cored trees) was selected for stable-carbon isotope analyses. We selected cores from six trees in each unit (18 per treatment) and used a scalpel to section each ring from 1969 to 2015 into EW and LW samples. The cores were selected at randomly from the set of cored trees within each unit after excluding trees that established after 1930 to avoid potential distortion of isotope signals early in our chronologies due to the "juvenile effect" (Leavitt, 2010; McCarroll & Loader, 2004). We also excluded trees with growth patterns that were uncharacteristic compared to the mean BAI pattern for the treatment.

We pooled the samples from the six trees per unit (Leavitt, 2008), providing one EW and one LW carbon-isotope chronology for each unit, and three EW and LW chronologies per treatment (control, thin only, wet burn, and dry burn). The chronologies include the treatment year (1992) plus each year over the 23-year windows before and after treatments (1969–1991 and 1993–2015). After pooling EW and LW samples by year for each unit, we used a Wiley Mill (Thomas Scientific, Swedesboro, NJ) to grind the samples into fine shavings and heat-sealed them in pouches of 25-micron filter paper (ANKOM, Fairport, NY). Then we extracted the waxes, resins, and oils that are potentially mobile across ring boundaries following Leavitt & and Danzer

(1993). We ground the samples to homogenize them to a fine powder before sending them to the Stable Isotope Laboratory at Washington State University (Pullman, WA).

Stable-carbon isotope composition (the ratio of 13C to 12C) relative to the known Vienna Pee Dee Belemnite (VPDB) standard is expressed as  $\delta$ 13C (µmol mol–1) = (Rsample / Rstandard – 1) × 1000. To remove trends in  $\delta$ 13C due to rising atmospheric CO2 concentration, we converted the 13C/12C ratios to the discrimination against 13C during carbon fixation. Carbon-isotope discrimination ( $\Delta$ 13C) represents the difference in  $\delta$ 13C between the air ( $\delta$ 13Cair) and the plant ( $\delta$ 13Cplant) due to the preferential use of 12C during photosynthesis (Farquhar, Ehleringer, & Hubick, 1989). Following McCarroll and Loader (2004),  $\Delta$ 13C (‰) = ( $\delta$ 13Cair –  $\delta$ 13Cplant) / (1 –  $\delta$ 13Cplant /1000). For  $\delta$ 13Cair, we used the values provided in McCarroll and Loader (2004) and supplemented more recent values with the annual mean of the monthly values recorded at Mauna Loa, Hawaii (http://scrippsco2.ucsd.edu/data/atmospheric\_co2/mlo).

Carbon-isotope discrimination may also be calculated as  $\Delta 13C$  (‰) = a + (b – a) (ci / ca), where ci and ca are the CO2 concentrations of leaf intercellular space and the ambient air, respectively, a is the discrimination against 13CO2 during diffusion through the stomata (–4.4‰), and b is the net fractionation due to carboxylation by Rubisco (–27‰) (McCarroll & Loader, 2004). This relationship illustrates that  $\Delta 13C$  is driven primarily by the ratio ci to ca, which reflects differences in the rates of photosynthesis and stomatal conductance. When stomatal conductance is higher than photosynthesis (typically when plants face little climatic stress), ci increases relative to ca, leading to carboxylation discrimination against 13C and higher  $\Delta 13C$ . With increasing climatic stress, plants typically close or reduce their stomatal openings, reducing both ci / ca and  $\Delta 13C$ .

## **Historic Lick Creek Photopoints**

We utilized a professional photographer (Mr. Dennis Simmerman) to travel to Lick Creek Research/Demonstration Area and collect current photographic images of forest structure at the network of historic photopoints first established in the early 1900s and periodically re-shot during the 20<sup>th</sup> century. Mr. Simmerman had conducted the prior photo set and was familiar with plot locations and photography protocols. He identified camera shooting angles and camera settings that replicated historic conditions, limiting phototaking to those periods where environmental conditions provided optimum lighting consistent with historic images. The condition of plot monumentation at each photopoint was noted, as well as any maintenance needs. Photos were taken in fall 2015, and again in fall 2016 – after the surrounding area had been treated by a landscape scale fuels mitigation project conducted by the Bitterroot National Forest.

#### **Results and Discussion**

#### **Biomass Recovery**

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 (Figure 4). 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 (Figure 4), 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.



**Figure 4.** 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.

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 <sup>-1</sup> (WB) to 30.3 Mg ha <sup>-1</sup> (DB), the NB treatment had 118.0 Mg ha <sup>-1</sup> 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 burning treatments.

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. In the thinning, lower forest floor biomass was due to less duff and litter in each of the two burned treatments relative to the control. Litter also tended to be lower in the NB units relative to the control. 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 artefact of the underburning, even 23 years after treatment. In the shelterwood, only duff was lower for both burned treatments relative to the control. CWD also tended to be lower in the shelterwood WB relative to the control, but not the DB. 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.

Snag volumes in treated stands were either the same as controls (thinning), or drastically less than 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.).

Understory vegetation is homogeneous across treatment types 23 years after treatment. When fuel treatments in fire-frequent forest types involve overstory tree removal, understory production typically increases with improved access to sunlight and belowground 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, 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. 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. At Lick Creek, all three fuel reduction treatments at 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.

#### **Fuel Loading**

In 2015, there were a few key differences in the surface fuel components among treatments in both installation. In the thinning, lasting effects of the treatments were observed only among fine fuels. One-hour fuels and duff were 67% and 78% lower in the SB relative to CO, respectively, while litter was 19-28% lower in all treated units relative to control (Figure 5). This is explained by the fact that there are fewer trees in the treated units to cast needles compared to the control.



**Figure 5.** Statistically significant differences in surface fuel loadings by treatment in 2015: One-hour fuels (a) and litter and duff (b) in the Thinning (left), and one-hour fuels (c) and duff (d) in the Retention Shelterwood (right). Bars denote mean fuel loading per particle type per treatment with +/- 1 standard error (n=3). Treatments are Control "CO", No Burn "NB", Cut+Spring Burn "SB" (Thinning only), Cut+Fall Burn "FB" (Thinning only), Cut+Wet Burn "WB" (Shelterwood only), and Cut+Dry Burn "DB" (Shelterwood only). Uppercase letters denote significant differences at  $\alpha$ = 0.05, while lowercase letters denote significant differences at  $\alpha$ = 0.10. In the shelterwood, the WB and DB units had 55% lower duff loading but there were no differences in litter layer by treatment. One-hour fuels were 62-87% lower in treated units in the shelterwood. Stands opened by harvesting exhibit less self-pruning and therefore drop fewer branches to the forest floor. Additionally, increased sunlight exposure and precipitation throughfall to the forest floor in open stands increases surface fuel decomposition rates (Keane 2008).

Harvesting operations inevitably generate some residual slash that affect surface fuel loading and fire behavior. When whole-tree harvesting is utilized, thinning operations have negligible effect on surface fuels. However, when limbs and tree tops are left on site, whether masticated or scattered, the residual slash increases wildfire intensity (i.e. fireline intensity). In our case, whole trees were harvested but tree tops above 15.24 cm diameter were left on site after felling. Tree limbs from the merchantable bole were removed and pile-burned near roadsides, reducing the potential amount of activity fuels (Arno 1999a). The residual tree-top slash was either consumed in the prescribed burns or largely decomposed to rotten material by 2015.

Accordingly, there were few long-term responses of surface fuel loading between post-treatment and 2015. Across installations, FWD surface fuels were lower or had returned to post-treatment levels. CWD was 3-4 times higher in the shelterwood installation in both prescribed fire treatments but the majority of that CWD was made up of rotten material, likely from post-treatment activity fuels (i.e. tree tops left on site that did not burn) and mortality caused by the prescribed fires. The composition of the litter/duff layer changed from mostly duff pre-and post-burning to mostly litter in 2015. However, because we have so few repeated measures of the fuel profile since the treatments, it is not possible to determine whether the high and low extremes of surface fuel loading over time were captured. Additionally, since there was no pre-harvest measurement of all the fuel components, it is impossible to distinguish between the impacts of harvesting versus prescribed burning.

By 2015, the thinning, which had few saplings to begin with, experienced increases of 1.5-4 times the amount of saplings since post-treatment, while the treated units in the shelterwood had increases of 28 to 362 times the post-treatment levels (Table 1). In both installations, the larger residual trees increased in size but only the thinning maintained a unimodal diameter class distribution. In the shelterwood, the smallest diameter classes saw an increase of small trees, mostly Douglas-fir. Retention shelterwood cuttings are intended to facilitate regeneration while retaining large trees of the desired species. While the treatments intended to recruit ponderosa pine, mature Douglas-fir overstory existed on site as a seed source for natural regeneration. Additionally, advance regeneration survived the treatments, especially in cut, but unburned, units. Arno (1999b) previously reported at Lick Creek that five-years post-treatment, Douglas-fir advance regeneration (seedlings and saplings >5 years old) averaged over 3200 stems ha<sup>-1</sup> in the NB units compared to 0 stems ha<sup>-1</sup> in cut-and-burn (WB and DB) treatments. Even posttreatment Douglas-fir regeneration (seedlings <5 years old) exceeded 760 stems ha<sup>-1</sup> in the cutonly units, more than 3-5x higher than cut-burn units. In all treated units, post-treatment ponderosa pine regeneration averaged 770 stems ha<sup>-1</sup>. Whereas prescribed burning eliminated seedlings and saplings, they were not eliminated by the harvesting alone (i.e., as seen in the NB units). In this instance, a shelterwood system without additional surface fuel treatments, while successfully promoting regeneration, benefited the shade-tolerant species most and undermined

the long-term treatment goal of increasing ponderosa pine as the dominant second age class across the installation (Arno 1999a).

The emergent ladder fuels translated directly to increased canopy fuel variables. In the thinning, canopy fuel loading and canopy bulk density were consistently ~30% lower in treated units, regardless of season of burn. In the shelterwood, burned units (WB and DB) maintained lower Douglas-fir seedling and sapling regeneration compared to NB units, which translated directly to lower canopy fuel loading in this lower stratum.

## **Understory Vegetation**

Restoration treatments resulted in temporary reductions of understory cover (except forbs), but they returned to pre-treatment levels in 15 years after harvesting. Species richness increased and then declined, but remained slightly higher than pretreatment. Regardless of origin, forbs showed the strongest responses to the restoration treatments. Overstory basal area was negatively associated with understory cover, but there were no significant associations with species richness and composition. Results demonstrate understory vegetation of this forest type was resilient to the treatments in the long term.

In general, understory vegetation recovered rapidly -- within 3 years after treatment. Burned treatments, especially, exceeded pre-harvest cover just 5 years after treatment due to the increased growing space and resource availability released by fire-caused mortality of competitors. However, the initial vegetation flush faded over time, and elevated understory vegetation cover returned to pre-harvest levels 15 years after treatment. We found that the NB treatment had less total understory cover 23 years after treatment than pre-treatment; this recent decline is presumably due to the continuous growth of tree regeneration displacing the understory.

Of all life forms, forbs seem to benefit most from restoration. The restoration treatment immediately and drastically increased forb cover, and greater disturbance intensity (i.e., WB and DB compared to NB) resulted in even higher forb cover. Restoration treatments that included prescribed fire were especially beneficial because burning removed surface organic matter such as litter and woody debris, creating favorable environmental conditions for forb germination. These conditions gave forbs the advantage until the other life forms (i.e., shrubs and graminoids) re-established or re-colonized.

In addition, the positive forb response to treatment is further explained by higher initial forb species richness. Although forbs occupied the least space (i.e., cover) of all life forms prior to treatment, forb richness was highest. It appears that the forest floor seedbank was profuse with forb species that were readily stimulated by the burning treatments. Overall, these results indicate that forbs play a critical role in understory revegetation following treatment.

Similar to the trends we observed in vegetation cover, species richness increased after restoration (Figure 6), and forbs drove the overall changes in richness. In addition, harvesting followed by burning increased species richness beyond the unburned NB treatment, because burning favored

(a) 🔏 (b) Harvest and Burning Harvest and Burning 3 Species Richness Species Richness 12 10 Control 0 Control Cut and dry burn ٠ ٠ Cut and dry burn □ Cut only □ Cut only Cut and wet burn Cut and wet burn 0 +23+15 +5 +15 +23 +0+3+3+5 -1 -1 +0Years after Harvest Years after Harvest (c) a (d) 🤟 Harvest and Burnin Harvest and Burning 0 Control Cut and dry burn Cut only Cut and wet burn 12 Species Richness Species Richness 10 ф Control ٠ Cut and dry burn □ Cut only Cut and wet burn 0 C +15+23+3 +15+23 +0+3+5+0+5-1 -1 Years after Harvest Years after Harvest

understory proliferation by releasing soil nutrients, improving moisture and light conditions, removing competitors, and creating open spaces for seed dispersal and germination.

**Figure 6.** Changes in species richness for (a) total understory vegetation, (b) shrub, (c) forb, and (d) graminoid by time. Data are represented prior to treatment (-1), immediately after treatment (+0), +3, +5, +15, and +23 years after treatment, respectively. Error bar indicates 1 standard error of the mean.

Our results demonstrated that overstory basal area was negatively associated with both understory cover and species richness. Overstory trees are dominant competitors over understory vegetation, appropriating space, light, nutrients, and water. Although understory vegetation responses to competition are species-specific, resource availability generally limits understory response. Therefore, we expect that understory cover and species richness increases as overstory retention decreases, but the magnitude and rate of understory responses vary with environmental conditions and species composition. Our CCA and permutation tests indicated that each treatment created distinctive communities in time. Throughout the measurement years, plot-scale species composition was quite heterogeneous; there was no predominant (i.e., exceeding 10% of understory cover) understory species across the treatments. Thus, treatment distinctiveness was attributed to different low-coverage species assemblages rather than high-cover dominance by select species. Because these treatments were implemented at the same time and overstory basal area was not associated with understory species composition, differences in species composition were presumably caused by treatment type and slight differences in initial (pre-existing) species composition.

We observed parallel compositional shifts in both burning treatments on the CCA projection, implying similar species composition changes. Despite the different moisture conditions in the treatment burning windows, our WB and DB treatments were only implemented two weeks apart, therefore any treatment differences would be attributable to differences in burn consumption, not vegetation phenology. Duff consumption (from depth measurements) was 2.2 times greater in the DB than WB, demonstrating the greater prescribed fire intensity and impact in the former treatment. Nevertheless, fire intensity did not have a significant effect on understory species composition, suggesting that the species responding at Lick Creek can thrive with varying degrees of surface fire.

Immediately after the treatments, we found distinctive fire-stimulated, re-sprouting species in the burned units (e.g., *Silene menziesii* Hook., *Claytonia perfoliata* Donn ex Willd., and *Apocynum androsaemifolium* L.), whereas a non-sprouting plant species that could not survive burning (i.e., *Trisetum spicatum* (L.) K. Richt.) was exclusively found in the NB treatment. Shortly after burning treatments, colonizers such as *Epilobium brachycarpum* C. Presl, *Penstemon* Schmidel, *Hieracium* L., and non-native species likely seeded in and established from unburned sites. A decade and a half after treatment, plant species with medium to high fire tolerance (e.g., *Carex* spp., *Rosa gymnocarpa* Nutt., and *Lupinus sericeus* Pursh) characterized the treated areas. Overall, the results indicated that fire intensity in our study was not enough to eliminate resprouting plant species, and that succession only partially followed the expectation that species with seedbank and rapid dispersal strategies dominate early post-disturbance stages, whereas species with vegetative reproduction strategy prevail in later seral stages.

Our result that post-harvest burning increased non-native species richness over the NB treatment implies that the combination treatments represent an intermediate level of disturbance, which is consistent with other forest cutting and burning studies. Forest managers should be aware that restoration with burning treatments also benefits non-native plant establishment by providing available resources, scarifying seedbeds, and creating openings.

Our results indicated that non-native species responded to restoration treatments just as forbs did (Figure 7), precisely because the majority of non-natives in the study were forbs (27 out of 36 species). In addition, non-native richness was positively correlated with overall understory (Pearson's correlation coefficient r=0.34, P<0.001) and native plant species richness (r=0.33, P<0.001). These results indicate that both native and non-native plant species take advantage of high resource availability after restoration treatments. The minor and temporary spike in non-native cover and richness following restoration in this study is comparatively benign, and our

findings advocate that restoring fire-prone forests and reducing fire hazard benefits native flora conservation.



**Figure 7.** Temporal (a) cover (%) and (b) species richness changes of non-native species after restoration treatment. Data are represented prior to treatment (-1), immediately after treatment (+0), +3, +5, +15, and +23 years after treatment, respectively. Error bar indicates 1 standard error of the mean.

Overstory retention reduces available understory resources such as light, moisture, and nutrients. The result that non-native plant cover and species richness were not associated with the basal area of retained overstory trees may indicate that non-native plants are better competitors than native plants. Native plants had greatest cover and richness with low overstory densities, but non-native plants thrived amid all overstory densities. This demonstrates a tradeoff between overstory retention and native understory cover and richness. Although some overstory presence may act as a colonization barrier and buffer against overwhelming non-native plant invasion, limiting overstory retention may improve competitiveness of native understory species in this forest type. This finding is especially important for managers that aim to improve resistance to wildfire, because it indicates that managing for high native understory plant establishment coincides with the low overstory retention in fuel reduction treatments to reduce potential crown fire spread.

#### Resilience

Restoration treatments had substantial effects on tree growth and physiology that persisted for 23 years. The reduction of competition altered trajectories of tree growth and  $\Delta^{13}$ C by contributing to a sustained growth release (Figure 8) and amplifying the intra-annual variation in  $\Delta^{13}$ C by increasing EW  $\Delta^{13}$ C and decreasing LW  $\Delta^{13}$ C relative to controls. These responses were similar

for all treatments other than a more gradual increase in growth and EW  $\Delta^{13}$ C in burned than unburned treatments of the shelterwood.

We found little difference in climate–growth relationships between treated and control chronologies other than a slight reduction in LW growth sensitivity to late-summer climate in the two cut and burn treatments of the shelterwood. This difference likely had little influence on tree growth given that LW accounts for a small portion of annual growth and relationships to winter precipitation remained strong in all treatments. The treatments had little effect on EW  $\Delta^{13}$ C sensitivity to climate. However, we found a substantial increase in LW  $\Delta^{13}$ C sensitivity that was similarly strong in burned and unburned treatments.

The implications of these changes for reducing tree vulnerability to drought and drought-related stresses became evident over the second post-treatment plot census interval (2005–2015), during which mountain pine beetle activity increased locally, and the tree mortality rate for the control units of each experiment increased to more than double that of the respective treatments.

Our finding that after treatments EW and LW  $\Delta^{13}$ C changed in opposite directions relative to the control (i.e.,  $\Delta^{13}$ C increased in the EW and decreased in the LW) illustrates the importance of evaluating *intra*-annual variation in tree-ring stable-carbon isotope signals. If we had analyzed  $\Delta^{13}$ C responses in whole tree rings, we would have interpreted that cutting had minimal effect, or we would have attempted to account for the few minor differences among treatments (e.g., small differences in iWUE between burned and unburned treatments from 1994 to 1998). However, our interpretation inevitably would have been inconsistent with the more detailed insight we gained by separating EW and LW.

Because EW comprises the majority of annual growth, the post-treatment increase in BAI under all treatments was driven primarily by an increase in EW growth (EWAI) relative to the controls (Figure 8). LW forms under lower soil moisture and higher evaporative demand, but LW growth (LWAI) kept pace with the increase in EW under all treatments. In fact, the annual proportion of LW (PLW) increased for about a decade (Figure 8). This ability of LW to keep up with the increase in EW growth has important implications for understanding how the treatments altered tree physiology and growth in the face of climatic stress.

The pre-treatment pattern of higher  $\Delta^{13}$ C (lower iWUE) in EW than LW is consistent with a strategy to maximize C assimilation at the expense of water loss when water is readily available early in the growing season. For all treatments, the post-treatment increase in EW growth relative to the controls (Figure 8 c, d) coincides with the increase in EW  $\Delta^{13}$ C (decrease in EW iWUE), suggesting the EW growth release was driven in part by increases in stomatal conductance ( $g_s$ ) and leaf-level photosynthetic rates (A), where the increase in  $g_s$  outweighed that in A. These interpretations are consistent with field measurements of A and  $g_s$  in 2001 and 2002 under the thinning (Sala et al., 2005). The field data also show that foliage mass per tree increased following treatments, which further contributes to the increase in EW growth.



**Retention Shelterwood** 



Figure 8. Growth responses to cutting (1992) followed by broadcast burning (1993 & 1994): basal area increment (BAI; a, b), earlywood area increment (EWAI; c, d), latewood area increment (LWAI; e, f), and the proportion of latewood (PLW; g, h). Values are means +/- 1 standard error of the mean. Dashed vertical lines indicate the year of cutting (1992) plus an earlier cutting treatment (1967) preceding this study and conducted in portions of Thinning installation area.

The post-treatment increase in foliage produced under favorable moisture conditions early in the growing season could leave trees more vulnerable to drought as they need to continue to supply water to larger crowns when evaporative demand increases and soil-water becomes more limiting in late summer. At first this interpretation seems supported by our findings that LW  $\Delta^{13}$ C decreased and the relationship between LW  $\Delta^{13}$ C and late-summer climate became steeper

(i.e., a given level of climatic stress more strongly reduced LW  $\Delta^{13}$ C) after all treatments but not in the control. However, this interpretation is not consistent with our finding of increased LW growth after treatments (Figure 8 e, f), nor with previous findings that trees in all treatments of the thinning had higher July predawn water potential (suggesting lower whole-tree water stress) than control trees, and they maintained higher A and  $g_s$  than the controls from late June through late August (Sala et al., 2005).

Rather than being indicative of increased foliage area leaving trees at greater risk of late-summer drought, we suggest that the reduction in LW  $\Delta^{13}$ C and the strengthening of the relationship between LW  $\Delta^{13}$ C and climate after treatments indicates that trees in treated stands were able to fix C and incorporate it into new stem growth under more severe climatic stress than trees in untreated stands. For *Pinus* species, leaf gas exchange may be minimal in most summers because high evaporative demand and intense competition for limited water forces stomatal closure. The weaker relationship between LW  $\Delta^{13}$ C and late-summer climate before treatments and for the controls in the post-treatment period indicates that in denser stands, it took less severe climatic stress to substantially reduce or prevent new assimilation.

An intriguing finding is the short-term increase in the proportion of LW (PLW) observed from 1993 to about 2004 in all treated chronologies (Figure 8). Expansion of crown area and fine-root systems in response to reduced competition could explain this result. Because carbon allocation to stem growth is generally a lower priority than allocation to new foliage or fine roots, we might expect a short-term reduction in allocation to stem growth while trees re-adjust to the additional above- and belowground resources made available following cutting. If this biomass calibration occurred primarily while water availability was relatively high early in the growing season but not under drier conditions in late summer, we would expect reduced allocation to EW growth, and a corresponding proportional increase in LW growth. Eventually, as new foliage and fine-root growth equilibrated to the post-treatment growing conditions, the sink strength for new foliage and roots would have decreased and proportional allocations to EW and LW growth would have returned to pre-treatment levels (Figure 8 g, h). Interestingly, the time until PLW returned to pre-treatment levels roughly corresponds to the time it would take for a complete turnover of foliage (i.e., until all foliage produced before treatments was lost and replaced by foliage produced after the treatments).

The treatments enhanced resistance to at least one of the key drivers of ponderosa pine mortality (the mountain pine beetle), but the effects of treatments on tree mortality rates remained hidden for more than a decade until pressure from this mortality driver increased. All treatments of the thinning and shelterwood experiments maintained low tree mortality rates throughout the 23-year post-treatment analysis period. However, when local mountain pine beetle activity increased during the second post-treatment plot census interval (2005–2015), mortality rates in the controls of both experiments increased to more than twice that of their respective treatments. With mortality rates just under 2% yr<sup>-1</sup> in the controls, the increase in mountain pine beetle activity (primarily in 2011–2014) was not a severe outbreak. Yet, the ability of treated units to maintain lower mortality rates during this period indicates the treatments enhanced tree resistance to this important driver of ponderosa pine mortality.

One of the few differences between burned and unburned treatments was a more gradual growth increase in the wet burn and dry burn treatments compared to the cut only treatment of the shelterwood (Figure 8 b). However, there was no corresponding difference between burned and unburned treatments of the thinning (Figure 8 a). The sensitivity of LW growth to late-summer climate (precipitation, Tmax, and VPD) also decreased in the two cut and burn treatments, but not the cut only treatment of the shelterwood. This difference was largely due to differences in growth response over the first few years post-treatment, after which year-to-year variation in growth differed little among treatments.

The other main difference between burned and unburned treatments was a more gradual increase in EW  $\Delta^{13}$ C in the two cut and burn treatments compared to the cut only treatment of the shelterwood. This difference, along with the more gradual initial growth release in the cut and burn treatments, were both of short duration, and likely related to either fire-caused injuries (i.e., partial crown scorch) or alterations of soil nutrient cycling. Changes in inorganic nitrogen pools and nitrogen cycling rates were recorded in the first couple of years following burning in our study site (DeLuca & Zouhar, 2000; Newland & DeLuca, 2000), but the differences from unburned units were essentially lost by years 8–9 (Sala et al., 2005).

Our analyses support that the restoration treatments commonly applied in ponderosa pine forests are likely to provide some resistance to both drought and bark beetle disturbances. Specifically, trees in thinned stands were able to maintain physiological activity under greater climatic stress relative to trees in unthinned stands. However, we did not find that prescribed burning strengthens resistance to drought or bark beetles, at least under the intensity observed over the 23 years since treatment in our study area. However, because the burning killed nearly all pre-existing tree seedlings and saplings, stand infilling by post-treatment tree recruitment was slower in burned units compared to thin only units (Clyatt et al., 2017), which could lead to longer persistence of the treatment effects in burned units.

## **Science Delivery**

As part of our science delivery effort, this project produced the following outcomes:

- 5 peer-review journal articles (2 published; 1 in review; 2 in final stages of preparation)
- 1 conference proceedings article
- 2 MS theses
- 3 oral and 11 poster conference/workshop presentations
- 2 DOI-linked archived Forest Service datasets (1 published, 1 in press)
- 1 series of professional photopoint images to add to the historic Lick Creek set
- 1 field tour organized in coordination with the JFSP Northern Rockies Fire Science Exchange Network.

Many of those deliverables were produced or advanced by the two graduate students and three post-doctoral scientists that led key elements of analysis. Through education and/or work

experience, their professional careers were advanced with full or partial support of the JFSP as a result of their participation in this project.

We intend to bring our two articles in preparation to the submission stage for peer-review within the next three months. Once all original research articles have completed the peer review process and are in the publication stage, we intend to produce a comprehensive synthesis article that summarizes Lick Creek's past and most recent research findings.

# **Conclusions and Implications**

Fuels reduction treatments are common in ponderosa pine ecosystems of the interior western United States, but the long-term effects on many key ecosystem attributes remain poorly understood, including: tree growth and mortality; forest fuel loads; understory vegetation diversity and composition; production and distribution of aboveground biomass; and physiological response of trees to drought stress.

Among this study's findings, the following are most salient:

- Tree biomass subjected to fuels treatments recovers to pre-harvest levels in less than 23 years, while stands gain the benefit of reduced stand densities that promote forest restoration objectives. Across all treatments, tree biomass in this study recovered to pre-harvest levels by 2015 (after 23 years).
- Treatment specifics do not greatly influence seedling, vegetation, and stump biomass, which in this study were similarly affected by all treatments. Forest floor and snag biomass tends to be reduced by burning treatments, even after more than two decades.
- Treatments leads to a growth release that remains evident even after 23 years. In this study, reducing competition via restoration treatment enables trees to fix carbon and incorporate it into new stem growth, whereas slower-growing trees in uncut stands either cease new assimilation or become more dependent on stored carbohydrates.
- Restoration treatments improve tree survivorship. In this study, tree mortality rates in the controls were more than double the treated units.
- Treatments maintained substantially lower canopy fuel loads (and lower canopy bulk density in thinning units), and tended to lower 1-hr fuels, litter, and duff. The heavier cutting associated with the shelterwood accelerated ladder fuel development, reducing canopy base height. Understory cover (except forbs) was initially reduced by treatments, but then returned to pre-treatment levels within 15 years. Species richness increased, and then declined, but remained slightly higher than pretreatment. Forbs responded most strongly to treatments. Understory cover was negatively related to overstory basal area (but species richness and composition were unrelated to basal area).
- Fuel treatment longevity is strongly influenced by silvicultural prescription specifics. Despite common initial forest conditions, different treatment combinations produce different fuel loads and fuel structures over time.

- Stand density reduction is key to improving the ability of residual trees to tolerate climatic stress and associated biotic disturbances. Faster growth and enhanced ability to assimilate carbon under more stressful climate following treatments leads to greater tree survivorship.
- Understory vegetation is resilient to a variety of restoration and fuel treatments, including all of those represented here. However, the more severe the treatment the greater the deviation from pre-treatment and greater non-native understory vegetation.

Overall, this study showed the high utility of restoration and fuel treatments to promote a spectrum of management objectives and variety of ecosystem responses. Results indicate that fuels treatments are effective, important, and enduring; the treatment specifics are less important. Greatest differences were observed between treated and untreated units, and there were relatively few (and only moderate) differences among treatments. Near-term differences among treatments tended to abate over time.

Treatments performed during a short period more than 20 years have imparted attributes that continue to benefit those units in multiple ways. With some exceptions, effects of post-cutting broadcast burning were modest and attenuated more over time, indicating the need to repeat burning with a greater frequency than the single entry represented by this study.

This project has also demonstrated the high value of research that is established with an experimental design suited to long-term analysis, and the importance in maintaining the infrastructure at research installations like Lick Creek. One side effect of conducting this study is that plot monumentation at the Lick Creek installation, and the professional relationships that support it, have been revived and strengthened, positioning this study area to continue generating valuable information into the future.

## **Future Research**

Our study was retrospective in nature, in that we focused on capturing many changes that occurred in response to treatment over more than two decades. There remain questions that can be addressed – for example, we are curious as to whether treatment and/or time have imparted drought resistance to the regeneration cohort (e.g., is recent regeneration more drought-resistant than past regeneration; is control regeneration less drought resistant than that those within treatement units?). Overall, however, we are satisfied that this project has resulted in valuable research findings, refurbished study area monumentation, consolidated datasets, and revitalized professional relationships. Taken together, those outcomes position the study area to remain productive as an important long-term research installation into the future.

Looking forward, we intend to coordinate with Bitterroot National Forest staff on the design and implementation of another round of experimental treatments to this study's treatment units. Forest management is dynamic, not static, and long-term research that informs management must also be dynamic. To remain their relevance and utility, experiments must incorporate additional

treatments over time, so that they reflect authentic treatment regimes. The shelterwood study is now 26 years post-establishment without any additional treatment overlays. This duration represents the upper end of the historic fire return interval in ponderosa-pine dominated forests in the Northern Rockies. The Lick Creek thinning installation received a re-entry cutting treatment in 2016 (treatment units were thinned to 40-60 sq. ft/acre; control units were maintained as-is), but no additional burning has occurred.

Two of this projects PI's (Hood, Keyes) are in the process of renewing a Memorandum of Understanding among the Bitterroot National Forest, the Rocky Mountain Research Station, and the University of Montana to ensure continued protection and viability of the Lick Creek Demonstration/Research Forest. In several meetings and site visits with the Bitterroot National Forest silviculturist, we have discussed future maintenance treatments that would be enhance the relevance of the experimental area while maintaining its research integrity. Those discussions are still in the preliminary planning stage, and will involve more parties before they are defined, but re-entry treatments in the shelterwood installation would likely include: control (same units as initial study; no action), prescribed burn only, thinning to increase structural heterogeneity, and thinning followed by a prescribed burn. We are also planning for a prescribed burning treatment to some units in the commercial thinning installation. We plan to maintain the original plot layout, and continue monitoring as funding allows for both studies.

Our research thoroughly analyzed the Lick Creek thinning and retention shelterwood installations. Yet, these two installations were originally established not as a tandem but as part of a triad. Between them lies a third installation – concurrently designed and implemented by noted restoration silviculturist Dr. Carl Fiedler - to evaluate uneven-aged silviculture as another possible strategy to achieve ponderosa pine forest restoration and wildfire hazard reduction objectives. In conversations with the Bitterroot National Forest about our findings, staff have shared a strong interest in reviving the uneven-aged study installation. That uneven-age study includes a cut-only treatment, a cut-and-burn treatment, and a no-action control. We didn't include that study in our JFSP-funded project because we were unable to locate the original data. In the process of documenting the thinning and shelterwood installations for remeasuring and archiving, we discovered the forest inventory and fuels data for the uneven-aged study units. We were also able to locate the treatment units and sampling monumentation in the field, and have confirmed that although the sampling method differs from the thinning and shelterwood installations, the treatment design is very similar, and remeasurement is possible. We see immense value in remeasuring that third study to complement the other efforts at Lick Creek. and are currently looking for ways to fund this remeasurement and archiving of that data.

Additionally, we plan to continue retaking photoplots of the Lick Creek drainage approximately every 10 years in order to maintain and add to that historic photoseries, which dates back to 1909. That legacy is an important one, and one that we wish to ensure continues.

## **Literature Cited**

- Arno SF. 1976. The historical role of fire on the Bitterroot National Forest. Research Paper INT-RP-187. Ogden, Utah: U.S. Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station. 29 p.
- Arno, S.F. 1999a. Ecosystem-based management treatments. In: Smith, H.Y., Arno, S.F. (Eds.), Eighty-Eight Years of Change in a Managed Ponderosa Pine Forest. USDA Forest Service Gen. Tech. Rep. INT-GTR-23. pp. 22-25.
- Arno, S.F. 1999b. Tree regeneration. In: Smith, H.Y., Arno, S.F. (Eds.), Eighty-Eight Years of Change in a Managed Ponderosa Pine Forest. USDA Forest Service Gen. Tech. Rep. INT-GTR-23. p. 35.
- Arno S.F., and C.E. Fiedler. 2005. Mimicking Nature's Fire Restoring Fire-Prone Forests in the West. Island Press: Washington, D.C.
- Brown, J.K. 1974. Handbook for inventorying downed woody material. General Technical Report INT-GTR-16. Ogden, Utah: U.S. Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station. 24 p.
- Brown J.K., R.D. Oberhau, and C.M. Johnston. 1982. Handbook for inventorying surface fuels and biomass in the Interior West. General Technical Report INT-GTR-129. Ogden, Utah: U.S. Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station. 52 p.
- Campbell, J., G. Alberti, J. Martin, and B.E. Law. 2009. Carbon dynamics of a ponderosa pine plantation following a thinning treatment in the northern Sierra Nevada. Forest Ecology and Management. 257:453-463.
- Caratti, J.F. 2006. FIREMON database user manual. USDA Forest Service Gen. Tech. Rep. RMRS-GTR-164-CD. 38 p.
- Clyatt, K.A. 2016. Long-term impacts of fuel treatments on tree growth and aboveground biomass accumulation in ponderosa pine forests of the Northern Rocky Mountains. M.Sc. thesis. University of Montana, Missoula, USA. 118 p.
- Clyatt, K.A., C.R. Keyes, and S.M. Hood. 2017. Long-term effects of fuel treatments on aboveground biomass accumulation in ponderosa pine forests of the northern Rocky Mountains. Forest Ecology and Management 400:587-599. doi: 10.1016/j.foreco.2017.06.021
- Connell, B.R., and J.G. Smith. 1970. Response of understory vegetation to ponderosa pine thinning in eastern Washington. Journal of Range Management 23(3):208-212.
- Cousins, S.J.M., J.J. Battles, J.E. Sanders, and R.A. York. 2015. Decay patterns and carbon density of standing dead trees in California mixed conifer forests. Forest Ecology and Management 353:136-147.
- Cybis Elektronik & Data AB. 2018a. CDendro Cybis dendro dating program (Version 9.2). Saltsjöbaden, Sweden.

- Cybis Elektronik & Data AB. 2018b. Cybis CooRecorder Image Coordinate Recording program (Version 9.2). Saltsjöbaden, Sweden.
- Daubenmire, R., 1959. A canopy-coverage method of vegetational analysis. Northwest Science 33:43-66.
- De Cáceres, M. and Jansen, F., 2016. indicspecies: Relationship between species and groups of sites. R package version 1.7.6.
- De Cáceres, M., and P. Legendre. 2009. Associations between species and groups of sites: indices and statistical inference. Ecology 90(12):3566-3574.
- De Cáceres, M., P. Legendre, and M. Moretti. 2010. Improving indicator species analysis by combining groups of sites. Oikos 119(10):1674-1684.
- DeLuca, T.H., and K.L. Zouhar. 2000. Effects of selection harvest and prescribed fire on the soil nitrogen status of ponderosa pine forests. Forest Ecology and Management 138(1):263-271.
- Dufrene, M., and P. Legendre.1997. Species assemblages and indicator species: The need for a flexible asymmetrical approach. Ecological Monographs 67(3):345-366.
- Farquhar, G.D., J.R. Ehleringer, and K.T. Hubick. 1989. Carbon isotope discrimination and photosynthesis. Annual Review of Plant Physiology and Plant Molecular Biology 40:503-537.
- Gendreau-Berthiaume, B., S.E. Macdonald, J.J. Stadt, and R.J. Hnatiuk. 2015. How dynamic are understory communities and the processes structuring them in mature conifer forests? Ecosphere 6(2):1-49.
- Gruell, G.E., W.C. Schmidt, S.F. Arno, and W.J. Reich. 1982. Seventy years of vegetative change in a managed ponderosa pine forest in western Montana-Implications for resource management. USDA Forest Service Gen. Tech. Rep. INT-GTR-130. 42 p.
- Harrington, M.G. 1999a. Stand structure response to harvesting and prescribed burning on shelterwood cutting and commercial thinning units. In: Smith, H.Y., Arno, S.F. (eds.), Eighty-Eight Years of Change in a Managed Ponderosa Pine Forest. USDA Forest Service Gen. Tech. Rep. INT-GTR-23. pp. 28-31.
- Harrington, M.G. 1999b. Prescribed burn weather, fuel moistures, and fuel reduction on all cutting units. In: Smith, H.Y., Arno, S.F. (eds.), Eighty-Eight Years of Change in a Managed Ponderosa Pine Forest. USDA Forest Service Gen. Tech. Rep. INT-GTR-23. pp. 27-28.
- Heyerdahl, E.K., P. Morgan, and J.P. Riser, II. 2008. Crossdated fire histories (1650 to 1900) from ponderosa pine-dominated forests of Idaho and western Montana. General Technical Report RMRS-GTR-214WWW. Fort Collins, Colorado: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 89 p.
- Holmes, R.L. 1983. Computer assisted quality control in tree-ring dating and measurement. Tree-Ring Bulletin 44:69-75.
- Jain, T.B., M.A. Battaglia, H.-S. Han, R.T. Graham, C.R. Keyes, J.S. Fried, and J.E. Sandquist. 2012. A comprehensive guide to fuel management practices for dry mixed conifer forests

in the northwestern United States. USDA Forest Service Gen. Tech. Rep. RMRS-GTR-292. 331 p.

- Leavitt, S.W. 2008. Tree-ring isotopic pooling without regard to mass: No difference from averaging δ13C values of each tree. Chemical Geology 252(1–2):52-55. doi: 10.1016/j.chemgeo.2008.01.014
- Leavitt, S.W. 2010. Tree-ring C–H–O isotope variability and sampling. Science of the Total Environment 408(22):5244-5253. doi: 10.1016/j.scitotenv.2010.07.057
- Leavitt, S.W., and S.R. Danzer. 1993. Method for batch processing small wood samples to holocellulose for stable-carbon isotope analysis. Analytical Chemistry 65(1):87-89. doi: 10.1021/ac00049a017
- Livingston, A.C., J.M. Varner, E.S. Jules, J.M. Kane, and L.A. Arguello. 2016. Prescribed fire and conifer removal promote positive understorey vegetation responses in oak woodlands. Journal of Applied Ecology 53(5):1604-1612.
- Lutes, D.C., R.E. Keane, J.F. Caratti, C.H. Key, N.C. Benson, S. Sutherland, and L.J. Gangi. 2006. FIREMON: The fire effects monitoring and inventory system. General Technical Report RMRS-GTR-164-CD. Fort Collins, CO. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 400 p.
- Lutes, D.C., E.D. Reinhardt, R.E. Keane, J.H. Scott, and L.J. Gangi. 2016. FuelCalc (version 1.4.0). Accessed May 21, 2017. Available at: https://www.firelab.org/project/fuelcalc.
- McCarroll, D., and N.J. Loader. 2004. Stable isotopes in tree rings. Quaternary Science Reviews 23(7):771-801. doi: 10.1016/j.quascirev.2003.06.017
- Menakis J.P. 1994. Summary of Ponderosa Pine (*Pinus ponderosa*) Management Activities in the Lick Creek Drainage of the Bitterroot National Forest. M.Sc. thesis. Univ. of Montana, Missoula, MT, USA.
- Mincemoyer, S. 2013. Checklist of Montana Vascular Plants: Dated June 3, 2013. Montana Natural Heritage Program, Helena, MT. 97 pp.
- Newland, J.A., and T.H. DeLuca. 2000. Influence of fire on native nitrogen-fixing plants and soil nitrogen status in ponderosa pine-Douglas-fir forests in western Montana. Canadian Journal of Forest Research 30(2):274–282.
- Oksanen, J., F.G. Blanchet, M. Friendly, R. Kindt, P. Legendre, D. McGlinn, P.R. Minchin, R.B. O'Hara, G.L. Simpson, P. Solymos, M.H.H. Stevens, E. Szoecs, and H. Wagner. 2017. vegan: Community Ecology Package. R package version 2.4-3.
- Pfister, R.D., B.L. Kovalchik, S.F. Arno, and R.C. Presby. 1997. Forest habitat types of Montana. USDA Forest Service Gen. Tech. Rep. INT-GTR-34. 174 p.
- R Core Team. 2018. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. URL https://www.R-project.org/.
- Raile, G.K. 1982. Estimating stump volume. USDA Forest Service Res. Pap. NC-RP-224. 4 p.
- Sala, A., G.D. Peters, L.R. McIntyre, and M.G. Harrington. 2005. Physiological responses of ponderosa pine in western Montana to thinning, prescribed fire and burning season. Tree Physiology 25(3):339-348.

- Scott J.H., and E.D. Reinhardt. 2001. Assessing crown fire potential by linking models of surface and crown fire behavior. Research Paper RMRS-RP-29. Fort Collins, Colorado: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 59 p.
- Smith, H.Y., and S.F. Arno (eds.). 1999. Eighty-eight years of change in a managed ponderosa pine forest. USDA Forest Service Gen. Tech. Rep. RMRS-GTR-23. 55 p.
- Standish, J.T., G.H. Manning, and J.P. Demaerschalk. 1985. Development of biomass equations for British Columbia tree species. Canadian Forest Service Pacif. For. Res. Centre. Info. Rep. BC-X-264. 48 p.
- ter Braak, C.J. 1987. The analysis of vegetation-environment relationships by canonical correspondence analysis. Vegetatio 69(1):69-77.
- ter Braak, C.J., and P.F. Verdonschot. 1995. Canonical correspondence analysis and related multivariate methods in aquatic ecology. Aquatic Sciences 57(3):255-289.
- USDA NRCS. 2017. Plants database. https://plants.usda.gov. Accessed 1 October 2017.
- Woodall, C.W., L.S. Heath, G. M. Domke, and M.C. Nichols. 2010. Methods and Equations for Estimating Aboveground Volume, Biomass, and Carbon for Trees in the U.S. Forest Inventory. USDA For. Serv. Gen. Tech. Rep. NRS-GTR-88. 34 p.

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# **Appendix B: Completed/Planned Publications & Science Delivery Products**

## **Peer-Reviewed Journal Articles**

## Published

Crotteau, J.S., S.M. Hood, D.C. Lutes, C.R. Keyes, A. Sala, and M.G. Harrington. 2018. Management and succession at the Lick Creek Demonstration/Research Forest, Montana. Journal of Forestry 116(5):481-486. DOI: 10.1093/jofore/fvy030.

Clyatt, K.A., C.R. Keyes, and S.M. Hood. 2017. Long-term effects of fuel treatments on aboveground biomass accumulation in ponderosa pine forests of the northern Rocky Mountains. Forest Ecology and Management 400:587-599. DOI:10.1016/j.foreco.2017.06.021.

## In Progress

Tepley, A.J., S.M. Hood, C.R. Keyes, and A. Sala. *In review*. Forest restoration treatments in a ponderosa pine forest enhance physiological activity and growth under climatic stress: A treering carbon isotope approach. Journal of Ecology.

Bowen, K.J., C.R. Keyes, S.M. Hood, and C. Seielstad. *In prep.* Longevity of ponderosa pine fuel reduction treatments at Lick Creek Research/Demonstration Forest in western Montana, USA.

Jang, W., J.C. Crotteau, C.R. Keyes, S.M. Hood, D.C. Lutes, A. Sala, D.E. Pearson, and Y.K. Ortega. *In prep.* Long-term understory vegetation responses to restoration treatments in a northern Rocky Mountains dry conifer forest.

## **Graduate Theses**

Bowen, K.J. 2017. Effectiveness and Longevity of Ponderosa Pine Fuels Reduction Treatments: A Legacy of Research at Lick Creek Demonstration/Research Forest in Western Montana. MS Thesis, University of Montana, Missoula, MT. 69 p. https://scholarworks.umt.edu/etd/11108.

Clyatt, K.A. 2016. Long-Term Impacts of Fuel Treatments on Tree Growth and Aboveground Biomass Accumulation in Ponderosa Pine Forests of the Northern Rocky Mountains (Lick Creek Demonstration/Research Forest). MS Thesis, University of Montana, Missoula, MT. 118 p. https://scholarworks.umt.edu/etd/10712.

#### **Conference/Symposium Proceedings Articles**

Bowen, K.J., C.R. Keyes, S.M. Hood, C. Seielstad, and D.C. Lutes. *In press*. Comparison of three methods for quantifying coarse surface fuel loading. In: Proceedings – The Fire Continuum Conference: Preparing for the Future of Wildland Fire. 21-24 May 2018, Missoula, Montana. USDA Forest Service Rocky Mountain Research Station RMRS-P-XXX.

#### **DOI-Linked Archived Datasets**

Hood, S.M., D.C. Lutes, J.S. Crotteau, C.R. Keyes, A. Sala, M.G. Harrington, and G.T. Munger. 2018. Lick Creek historic photograph series: A century of change in a ponderosa pine forest in western Montana, US. Fort Collins, CO: Forest Service Research Data Archive. DOI: 10.2737/RDS-2018-0023.

Hood, S.M., D.C. Lutes, C.R. Keyes, A. Sala, K.J. Bowen, M.G. Harrington, S.F. Arno, H.Y. Smith, D.E. Pearson, and Y.K. Ortega. *In prep*. Lick Creek Demonstration/Research Forest: 25-year fire and cutting effects on vegetation and fuels. A comprehensive, electronic Lick Creek Study dataset that is inclusive of all data collected at the site, from inception to present. Fort Collins, CO: Forest Service Research Data Archive. DOI: XX.X/XX-XX-XX.

#### **Conference/Symposium Oral Presentations**

Thinning and prescribed fire alter tree-level climate -growth relationships and reduce vulnerability to drought in ponderosa pine forests of the Northern Rockies. A.J. Tepley, S.M. Hood, A. Sala. American Geophysical Union. Washington, D.C. 10-14 Dec 2018. Oral presentation by A.J. Tepley.

Stand-density reduction treatments alter tree-level climate–growth relationships and vulnerability to drought in low-elevation ponderosa pine forests of the Northern Rockies. A.J. Tepley, S.M. Hood, and A. Sala. The Fire Continuum Conference: Preparing for the Future of Wildland Fire. Missoula, MT. 21-24 May 2018. Oral presentation by A.J. Tepley.

Longevity of fuel reduction treatments at Lick Creek Demonstration / Research Forest. K.J. Jenkins, C.R. Keyes, and S.M. Hood. 3rd Southwest Fire Ecology Conference and 1st Applied Fire Science Workshop – Beyond Hazardous Fuels: Managing Fire for Social, Economic, and Ecological Benefits, Tucson, AZ. 28 Nov – 2 Dec 2016. Oral presentation by K.J. Jenkins.

#### **Conference/Symposium Posters**

Comparison of two methods for quantifying coarse surface fuel loading. K.J. Bowen, C.R. Keyes, S.M. Hood and D. Lutes. The Fire Continuum Conference: Preparing for the Future of Wildland Fire. Missoula, MT. 21-24 May 2018. Poster presentation by K.J. Bowen.

Effectiveness and longevity of ponderosa pine fuels reduction treatments at Lick Creek Demonstration/Research Forest, Montana. K.J. Jenkins, C.R. Keyes, S.M. Hood, and A. Sala. Annual Meeting of the Montana Society of American Foresters, Missoula, MT. 14 Apr 2017. Poster presentation by K.J. Jenkins.

Long-term impacts of fuel treatments on aboveground biomass in ponderosa pine forests of the Intermountain West. K.A. Clyatt, C.R. Keyes, D.S. Page-Dumroese, and S.M. Hood. Annual Meeting of the Montana Society of American Foresters, Missoula, MT. 14 Apr 2017. Poster presentation by K.A. Clyatt.

Effectiveness and longevity of ponderosa pine fuel reduction treatments at Lick Creek Research/Demonstration Forest, Montana. K.J. Jenkins, C.R. Keyes, and S.M. Hood. 3rd Southwest Fire Ecology Conference – Beyond Hazardous Fuels: Managing Fire for Social, Economic, and Ecological Benefits. Tucson, AZ. 28 Nov – 2 Dec 2016. Poster presentation by K.J. Jenkins.

Effectiveness and longevity of ponderosa pine fuel reduction treatments at Lick Creek Research/Demonstration Forest, Montana. K.J. Jenkins, C.R. Keyes, and S.M. Hood. 2016 Society of American Foresters National Convention. Madison, WI. 2-6 Nov 2016. Poster presentation by K.J. Jenkins.

Effectiveness and Longevity of Ponderosa Pine Fuels Reduction Treatments at Lick Creek Demonstration/Research Forest, Montana. K.J. Jenkins, C.R. Keyes, and S.M. Hood. 2016 Annual Biomass Research and Development Initiative Meeting. Missoula, MT, 19 Apr 2016. Poster presentation by K.J. Jenkins.

Long-term Impacts of Fuel Treatments on Productivity and Biomass in Ponderosa Pine Forests of the Intermountain West. K.A. Clyatt, C.R. Keyes, D.S. Page-Dumroese, and S.M. Hood. 2016 Annual Biomass Research and Development Initiative Meeting. Missoula, MT, 19 Apr 2016. Poster presentation by K.A. Clyatt.

Effectiveness and Longevity of Ponderosa Pine Fuels Reduction Treatments at Lick Creek Demonstration/Research Forest, Montana. K.J. Jenkins, C.R. Keyes, and S.M. Hood. 5th International Fire Behavior & Fuels Conference. Portland, OR. 11-15 Apr 2016. Poster presentation by K.J. Jenkins.

Lick Creek Demonstration-Research Forest: 25- Year Fire and Cutting Effects on Vegetation and Fuels. S.M. Hood, C.R. Keyes, A. Sala, D.C. Lutes, K.A. Clyatt, and K.J. Jenkins. 6th International Fire Ecology & Management Congress: Advancing Ecology in Fire Management. San Antonio, TX. 16-20 Nov 2015. Poster presentation by S.M. Hood.

Long-term Impacts of Fuel Treatments on Productivity and Biomass in Ponderosa Pine Forests of the Intermountain West. K.A. Clyatt, C.R. Keyes, D.S. Page-Dumroese, and S.M. Hood. 2015 Society of American Foresters National Convention. Baton Rouge, LA. 3-7 Nov 2015. Poster presentation by K.A. Clyatt.

Effectiveness and Longevity of Ponderosa Pine Fuels Reduction Treatments at Lick Creek Research Forest, Montana USA. K.J. Jenkins, C.R. Keyes, and S.M. Hood. 2015 Society of American Foresters National Convention. Baton Rouge, LA. 3-7 Nov 2015. Poster presentation by K.J. Jenkins.

#### **Field Demonstration/Tours**

Forest Service field tour of the Lick Creek Demonstration/Research Forest. Part of the annual US Forest Service Northern Region Silviculturist/Wildlife Biologist Meeting – "Management on Dry Forest Habitat Types and Flammulated Owls." In coordination with the Bitterroot National Forest and the USFS Northern Region. 11 Jul 2017. Approximately 30 attendees (silviculturists, wildlife biologists, and forest planners) from the USFS and BLM.

#### Websites

<u>https://firelab.org/project/century-change-ponderosa-pine-forest</u> A Century of Change in a Ponderosa Pine Forest. Changes in forest structure over more than a century, as captured by the historic Lick Creek photopoints.

<u>https://firelab.org/project/ponderosa-pine-restoration-lick-creek</u> Ponderosa Pine Restoration at Lick Creek. Featuring the present JFSP-sponsored study, with description of study treatments and project analyses.

## **Other Outreach/Science Delivery**

Showcase of the Lick Creek study and historical photographs, US Forest Service booth at the Western Montana Fair, Missoula, MT. 8-12 August 2017. Approximately 1,000 visitors.

# **Appendix C: Data**

Data collected in this study include trees (all sizes on fixed-area plots), planar intercept for DWM, duff and litter depth, cover and height of live and dead herb and shrubs, logs (fixed-area plots), stumps (fixed-area plots), cover quadrats and macroplot cover. Digital photos are also included. Different methods were sampled at each visit and are described in the following files submitted to the Joint Fire Science Program:

- LickCreek\_CommercialThin\_Visits.docx
- LickCreek\_Shelterwood\_Visits.docx.

For each sampling method, data collected at the plot level is available in CSV format: 1) plot-byplot by visit and 2) data aggregated for all visits into one file. All data are also available in the original FFI database format. Digital photos are archived by date along with a description of the photo orientation. Data will be archived and made readily available at the US Forest Service Rocky Mountain Research Station Data Archive: https://www.fs.usda.gov/rds/archive/.