Susan D. Kocher[1] and **Daylin Wade,** University of California Agriculture and Natural Resources, Central Sierra MCP, 311 Fair Lane, Placerville, California 95667

Effects of Post-Fire Timber Harvest and Mastication on Shrub Regrowth in the Sierra Nevada Mountains: A Lake Tahoe Case Study

Abstract

Increasingly large high-severity wildfires in dry forests of the western United States have led to concern about how best to regenerate new forests after wildfires. Harvesting fire-killed trees, burning woody debris, and tree planting are commonly used reforestation strategies. This study evaluated the effects of a novel forest restoration approach that involved masticating un-merchantable dead trees and spreading the woody debris generated across the site to prevent erosion from the 2007 Angora Fire in Lake Tahoe, California. Woody material covered 82% of the site after treatment, with an average depth of 6.6 cm, and volume of 190 tonnes per hectare. We found that this treatment reduced shrub regrowth, compared to an untreated area nearby, and that shrub regrowth was inversely related to fuel depth. Seven years after the fire, shrub cover averaged only 50% on treated plots compared to 92% on untreated plots. The tallest shrubs averaged 69 cm in height on the treated site compared to 114 cm on nearby untreated sites. Tree seedlings planted on the treated site averaged 141 cm in height, well above the height of the shrubs. Advantages of this approach include controlling erosion while reducing drought stress, reducing the potential for weed introduction, and reducing the need for herbicide to control shrub competition. Although leaving a layer of woody material where new trees have been planted does constitute a fire hazard, so too does a vigorous shrub layer. Managers should consider and weigh these factors when deciding on a post-fire reforestation strategy.

Keywords: high-severity wildfire, post-fire management, salvage logging, woody mulch, reforestation

Introduction

The recent increase in high-severity fire in the conifer forests of the western United States is attributed to both fire suppression, which has led to an accumulation of forest fuels (Stephens and Fule 2005, Parks et al. 2018), and a warming climate, which leads to longer periods of high fire weather danger (Westerling et al. 2006, Keyser and Westerling 2017). This trend has been pronounced in the Sierra Nevada, where hundreds of thousands of hectares of conifer forests have burned in very large fires (Collins et al. 2019) and at high severity (Miller and Safford 2012, Safford and Stevens 2017) in the past few decades.

Recent research has found a lack of conifer regeneration after high-severity fire in dry mixed-conifer forests in California (Welch et al. 2016) and the Sierra Nevada (Tubbesing et al. 2020), ponderosa pine (*Pinus ponderosa* Lawson & C. Lawson) and Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco) forests in Colorado (Rother and Veblen 2016), and ponderosa pine forests in Arizona and New Mexico (Savage and Mast 2005, Ouzts et al. 2015). In California, high-severity fire affected natural regeneration by increasing shrub competition and the distance to live seed trees (Welch et al. 2016, Tubbesing et al. 2020).

Although early successional forests have ecological benefits (Swanson et al. 2011), concern has been growing that, without intervention, burned forests across large expanses may shift in composition of tree species (Tubbesing et al. 2020) or convert to other vegetation types altogether, such as shrub and grassland (Millar and Stephenson 2015, Coppoletta et al. 2016). Since these ecosystems typically sequester significantly less carbon, type conversion away from forested ecosystems is counter to many long-term goals held

[1] Author to whom correspondence should be addressed.
Email: sdkocher@ucanr.edu

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by state and federal agencies for climate change mitigation (Forest Climate Action Team 2018).

Although post-fire shrub habitat has been reported to benefit some wildlife species in western forests (Fontaine et al. 2009) and in the Sierra Nevada (Fogg et al. 2016), vigorous post-fire shrub recovery interferes with conifer seedling growth and survival through competition for light and water (Zhang et al. 2006, Stephens et al. 2020, Tubbesing et al. 2020). Post-fire dominance by regenerating shrubs can also reduce the richness of native herbaceous plants (Bohlman et al. 2016).

Reforestation is generally acknowledged to be an effective way to expedite re-establishment of conifers after a fire (Sessions et al. 2004). Although activities often involved in reforestation, including salvage logging and site preparation, may have negative impacts on a number of ecological values, including biodiversity (Lindenmayer and Noss 2006, Thorn et al. 2018), they have also been shown to reduce shrub cover in some cases (Knapp and Ritchie 2016). Shrubs regrow after fire through several different mechanisms (Pausas and Keeley 2014): 1) by establishing new plants that germinate from seeds in the soil seed bank; 2) through survival of existing burned plants that re-sprout by epicormic buds, lignotubers, rhizomes, or roots; or 3) by a combination of both germination and resprouting. The mechanisms by which shrub regrowth is inhibited by salvage logging will thus depend on shrub species and reproductive methods and the type or degree of logging-caused impacts.

In this study, we examined the effects of post-fire salvage logging conducted after a high-severity wildfire that burned in the watershed of Lake Tahoe in the Sierra Nevada, California in 2007. Since the operations took place in an environmentally sensitive watershed, salvage logging was followed by mastication and spreading of un-merchantable material across the site to reduce soil erosion hazard. Masticating and spreading woody debris after fire and salvage logging is not a common post-fire treatment in the Sierra Nevada region, but was carried out due to concern for Lake Tahoe, an Outstanding National Resource Water (California RWQCB 1995). Masticated material is commonly spread across fuel-reduction project sites around Lake Tahoe to reduce the potential for sediment flow into the lake and maintain clarity (Hatchett et al. 2006). The practice has been shown to be effective in reducing erosion throughout the basin, including in the fire area (Harrison et al. 2016).

We began monitoring forest recovery on the parcel scheduled for salvage logging, and a nearby control area one month after the Angora Fire. Two years after the fire, it became visually apparent that shrub regrowth was less vigorous on salvage-logged areas than nearby untreated areas. To investigate the causes, we studied the effect of fuel left by salvage logging and mastication on shrub growth in the burn area.

Methods

Study Area

The 2007 Angora Fire in South Lake Tahoe, California burned 1,255 ha of high-elevation (1,930 m) mixed conifer/eastside pine type forest composed of Jeffrey pine (*Pinus jeffreyi* Balf.), white fir (*Abies concolor* (Gord. & Glend.) Lindl. ex Hildebr), incense cedar (*Calocedrus decurrens* (Torr.) Florin), and sugar pine (*Pinus lambertiana* Douglas). Trees on site regrew after clear-cut logging in the 1880s to provide timber for Comstock lode mining in neighboring Nevada. The second-growth forest that developed in the area over the next 120 years had four times the density of original stands, which reconstructions show had about 120 trees per ha. There was also a pronounced shift away from Jeffrey pine dominance towards dense white fir and incense cedar in younger cohorts (Murphy and Knopp 2000).

Precipitation in the area is 60 to 75 cm annually, mostly falling as snow between November and April. Mean monthly temperatures range from 1 °C in January to 18 °C in July. Soils on site are derived from glacial and colluvial deposits in the Tallac gravelly coarse sandy loam series (Natural Resources Conservation Service 2020).

The 1,250 ha fire burned mostly at high severity, with 53% of its area experiencing over 75% mortality in the tree canopy (Carlson et al. 2009) (Figure 1). The burn area included lands under a

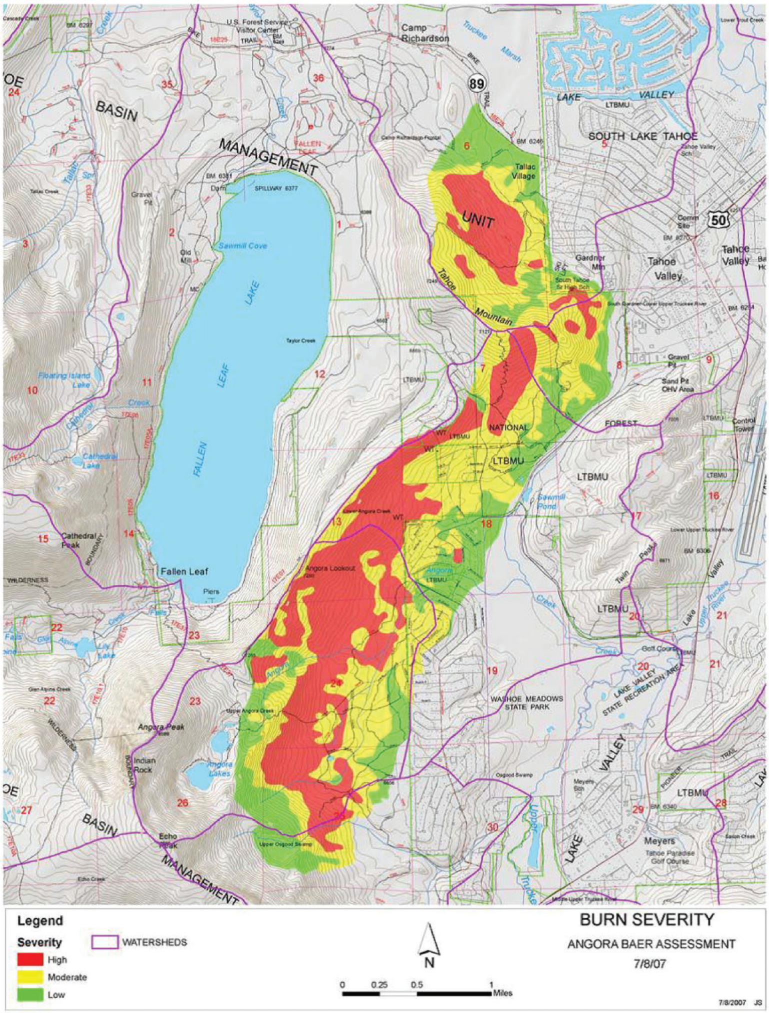

Figure 1. Location of the June 2007 Angora Fire. The fire burned south of Lake Tahoe, in the Sierra Nevada mountains, 300 km northeast of San Francisco, California. Burn severity of the Angora Fire (Weaver et al. 2007) shown as areas in red (high severity), yellow (moderate severity), and green (low severity).

mosaic of ownership and post-fire management strategies. Owners included the US Department of Agriculture Forest Service, the California Tahoe Conservancy, and private residential owners. We collected data on a 12-ha parcel owned by the Tahoe Conservancy where salvage logging and mastication was carried out in fall 2007, as well as on adjacent untreated lands managed by the Forest Service, which were also severely burned. Treatments on the salvage-logged and masticated parcel were carried out with a goal of expediting the development of a forest on site and reducing short-term erosion risk (Figure 2). Almost all merchantable trees (> 25 cm diameter at breast height [dbh]) were removed in fall 2007, except for an average of 15 snags per ha left for wildlife habitat. A masticator ground up smaller, unmerchantable trees and logging slash, which were spread widely across the site to reduce the potential for soil erosion and sediment delivery to Lake Tahoe. Container seedlings of Jeffrey pine, sugar pine and incense cedar were planted on the treated site in 2008, 2009, and 2010, totaling 320 per ha by 2014.

Hypothesis and Monitoring Methods

Based on previous studies, we hypothesized that salvage logging would affect post-fire recovery because of the large amount of ground disturbance involved compared to the nearby control area where no treatments were done. To measure this effect, we established six monitoring plots, 400 m^2 in size and 80 m apart, along a transect in 2007. On the adjacent untreated parcel, we established six control plots using the same methodology. Conditions on the treated and untreated plots were very similar. Both sites had a southeast aspect with slope between 10% and 18% throughout. Soil type was Tallac gravelly coarse sandy loam across the plots. Data we collected in 2007 showed that all plots were severely burned and no live trees remained following the fire. Stand development history and species

Figure 2. The post-fire treatment: a) fire-killed trees were skidded to a landing and taken to a lumber mill in September 2007; b) limbs and small trees (25 cm dbh and smaller) were masticated in October 2007 and left on site; and c) typical condition after treatment in November 2007.

composition was comparable, with Jeffrey pine and other mixed-conifer species present. The density of trees (larger than 10 cm dbh) per ha was somewhat greater on the untreated site (556), as compared with the treated site (391), or between 3 and 4 times greater than Murphy and Knopp's (2000) estimates of 120 trees per ha in the old-growth stands found in the area before logging in the 1880s.

The permanent plots were used to measure vegetation and fuels throughout the study period. We inventoried standing trees and any remaining vegetation and fuels one month after the fire on both sites, and again two months later after salvage logging occurred on the treated site. We then monitored treatment and control plots again in 2009 and 2014 to make comparisons between fuels and shrub growth on the treated and untreated sites.

Fuel Load (2007 to 2008)—In 2007, before salvage logging and mastication occurred, we surveyed fuels on all plots using the planar-intercept method (Brown 1974). We sampled fuels on two 11.3-m-long transects per inventory plot. The first transect was placed beginning at plot center, randomly chosen to run one direction or the other along the slope contour, and the other transect was randomly located 60° in one direction or the other from the first transect. Two months later, after merchantable dead trees were removed and the remainder were masticated, the site was covered with abundant fuels. We resurveyed these treatment fuels in 2008, but we did not use the planar-intercept method, since the method is not accurate for masticated fuelbeds. This is because fuels in the smallest size classes are numerous and extremely time-consuming and difficult to count (Kane et al. 2009). Instead, we used a modified method that involved collecting masticated material within a 2500-cm^2 frame laid along a Brown's transect (Kane et al. 2009). We collected then sorted the 1-hr and 10-hr

Figure 3. Forest condition on one plot on the treated site over time: a) July 2007, a month after the fire but before the tree removal and mastication treatment; b) summer 2010, three years after treatment showing some shrub and herbaceous growth; and c) summer 2014, seven years after the fire, dead trees in the background have fallen, and shrubs were taller and covered a greater area.

size classes (< 0.6 cm and 0.6 to 2.5 cm in diameter, respectively), they were then dried and weighed to yield measurements of 1-hr and 10-hr fuels. We used the planar-intercept method to quantify the 100-hr (2.5 to 7.6 cm) and 1000-hr (> 7.6 cm) fuels.

Vegetation (2007)—We used ocular estimates in July 2007 to identify live vegetation but found none. All surface vegetation, as well as woody debris and litter, were consumed by the fire on all plots (Figure 3). We revisited the site in November 2007, four months after treatment, and found no vegetation regrowth.

Hypotheses—By 2009, shrubs were growing on both sites, but were obviously less vigorous on the treated site. Based on this observation, we hypothesized that salvage logging and mastication after fire inhibits shrub regrowth. Given the drastically different fuel load on the treated and untreated sites in 2008, we then hypothesized that the depth of the layer of masticated woody material left on site influenced shrub growth in the following ways:

- Hypothesis 1: Fuel depth left behind by post-fire mastication has an inverse effect on shrub height.
- Hypothesis 2: Fuel depth left behind by post-fire mastication has an inverse effect on shrub cover.

Additional Monitoring Protocols—To examine these hypotheses, we added collection of vegetation and fuel depth data to our monitoring protocols. We established an additional four transects on each plot, using a tape stretching 15 m in each cardinal direction beginning at plot center. We placed a monitoring quadrat frame of 50 cm × 50 cm to the right of the tape at 3, 6, 9, 12, and 15 m, for a total of 20 monitoring quadrats per plot. These quadrats were used for monitoring shrubs and fuel depth in 2009 and 2014.

The number of plots surveyed varied by year due to logistics and, in one case, transformation of vegetation type due to a change in water flow after the fire. On the treated site, five plots were surveyed in 2009 and four were surveyed in 2014. On the untreated site, six plots were surveyed in 2009. In 2014, treatment had occurred on the majority of our control plots, so we were only able to collect data on two transects, one on each of two plots.

Shrub Cover and Height (2009, 2014)—Within each quadrat, we identified all shrubs by species and assessed cover of each species within the quadrat using cover categories by ocular estimate (Table 1). The tallest shrub of each species was measured to the nearest cm.

Fuel Depth (2009, 2014)—We were not able to use the modified Brown's transects to measure fuels along our vegetation transects because the method involves destructive sampling (i.e., removal of woody material and litter from the sampling frame which would affect future vegetation growth). Instead, we carried out a simple measurement of the depth of fuel within quadrats. Total masticated fuel depth (1-hr, 10-hr, and 100-hr) was measured to the nearest 0.25 cm in five locations (in each corner and the center) within each of the quadrats. Since size classes of fuels were well mixed, we measured the total depth, noting the presence of 1000-hr logs. Litter was difficult to separate from fuels and so was included in depth measurements.

TABLE 1. Vegetation cover categories. Vegetative cover by species was visually estimated within these cover categories.

Cover category	% Cover
0	0
0.5	> 0–1
3	> 1–5
10	> 5–15
20	> 15–25
30	> 25–35
40	> 35–45
50	> 45–55
60	> 55–65
70	> 65–75
80	> 75–85
90	> 85–95
97.5	> 95–100

Data Analysis

Fuel Quantity—All fuels data collected using the planar-intercept method in 2007 were used to calculate tonnes per ha by transect. Where the hybrid method was used in 2008, the dry weight of 1-hr and 10-hr fuels collected within the 2500-cm^2 frame was calculated in tonnes of fuel per ha within 1-hr and 10-hr size classes.

Shrub Height and Cover—A non-parametric Mann-Whitney test was used to compare shrub height and shrub cover on treated and untreated plots in 2009 and 2014.

Shrub Growth and Fuel Depth—For treated plots in each survey year (2009 and 2014), data sets were created for the tallest shrub (cm) of any species in each quadrat, the total cover of all shrub species per quadrat (%), and the greatest fuel depth (cm) per quadrat. Any fuel depth that included a 1000-hr log was removed from the analysis, since logs of that size are likely to block any vegetation growth (McGinness et al. 2010). A linear regression analysis was performed using the R statistical package (R Core Team 2016) to identify relationships between fuel depth and shrub height, and fuel depth and shrub cover. Fuel depth data were log transformed and shrub cover data were logit transformed to improve normality. All other assumptions of linear regression were satisfied.

Results

Fuel Quantity

On the treated site, surface fuel loads increased fourfold after salvage logging because masticated material was spread across the site. Surface fuel increased from an average of 47 tonnes per ha after the fire, with 98% in the largest size class, to an average of 190 tonnes per ha after logging and mastication ($P = 0.003$). Of that, 18% were 1-hr fuels, 24% were 10-hr fuels, 16% were 100-hr fuels, and 42% were in the 1000-hr size class. On the untreated site, we found an average of

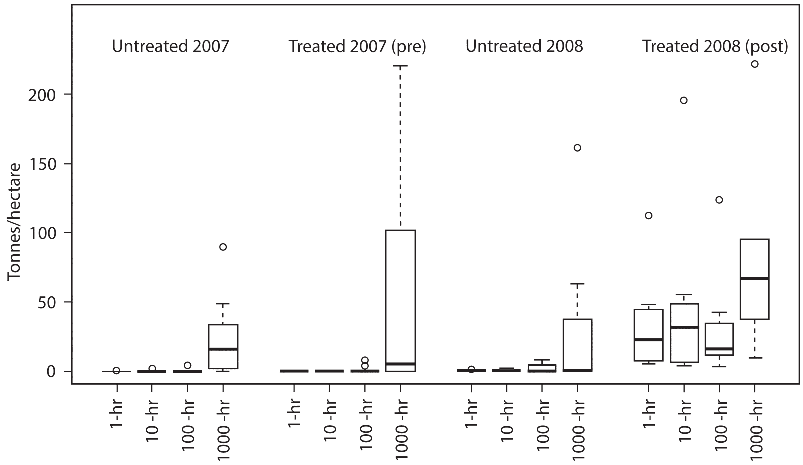

Figure 4. Surface fuel load by size class on the treated and untreated parcels in 2007 (pre-treatment) and 2008 (post-treatment). The dark horizontal line represents the median, and the bottom and top box edges represent the 25th and 75th percentiles. The 100-hr and 1000-hr fuel measurements come from fuel transects. For post-treatment data, 1-hr and 10-hr fuel classes were measured by collecting, sorting, drying, and weighing.

24 tonnes of fuel per ha in 2007 following the Angora Fire, with 97% in the 1000-hr size class. In 2008, we remeasured the fuel and found an average 28 of tonnes per ha, 89% in the 1000-hr size class, although neither difference was statistically significant (for total fuels $P = 0.84$ and for 1000-hr fuels $P = 0.26$) (Figure 4). The only significant difference we found in fuel size classes on the untreated site was an increase in 1-hr fuels, from 0.01 tonnes per ha directly after the fire in 2007 to 0.27 tonnes per ha in 2008 ($P = 0.002$).

Fuel Depth and Cover

Fuel was distributed across the site by the mastication process in a variable manner. Fuel was found at 82% of sampling points to an average depth of 3.7 cm in 2009 and at 97% of sampling points to an average depth of 5.5 cm by 2014. Variation was also high at the local, quadrat scale (2500 cm^2) where shrub growth was measured. All quadrats had fuel in at least one sampling point in each sampling year. In 2009, 54% of quadrats had at least some fuel at all five sampling points, and that percentage grew to 86% by 2014 as growing shrubs contributed leaf litter to the fuel layer. Most quadrats had at least one point with fuel at least 5-cm deep (53% in 2009 and 81% in 2014), but very few had fuel 5-cm deep on all five sampling points (9% in 2009 and 16% in 2014) (Figure 5). Because of variation in depths within quadrats, we chose the greatest depth of fuel in each quadrat as the independent variable when analyzing the effect on shrub growth. The greatest depth of fuel in each quadrat, including 1-hr, 10-hr, and 100-hr fuels, was an average of 6.6 cm in 2009 (standard error = 0.6 cm), and 8.3 cm in 2014 (standard error = 0.6 cm) ($P = 0.008$).

Shrub Cover

Shrub cover started at zero on both sites (as measured in 2007) and increased over the years, although at a faster rate on the untreated site (Figure 6). Shrub cover was significantly greater on the untreated site ($P < 0.001$) by 2009, averaging 28% ($n = 120$, standard error = 2.5), compared to 4% on the treated site ($n = 100$, standard error = 0.9). By 2014, shrub cover increased to an average of 50% on treated plots ($n = 80$, standard error = 3.7) and 92% ($n = 10$, standard error = 0.7) on untreated plots ($P = 0.001$). The most common shrub species encountered by far was the resprouting native mountain whitethorn (*Ceanothus cordulatus* Kellogg), common to the area. Also present were greenleaf manzanita (*Arctostaphylos patula* Greene), creeping snowberry (*Symphoricarpos mollis* Nutt.), Sierra currant (*Ribes nevadense* Kellogg), and Scouler's willow (*Salix scouleriana* Barratt ex Hook.). No non-native shrubs were found. Dominance by whitethorn increased over time on both treated and untreated sites (Figure 6a).

Shrub Height

Shrubs were significantly taller on the untreated site by 2009 ($P < 0.001$), with the average tall-

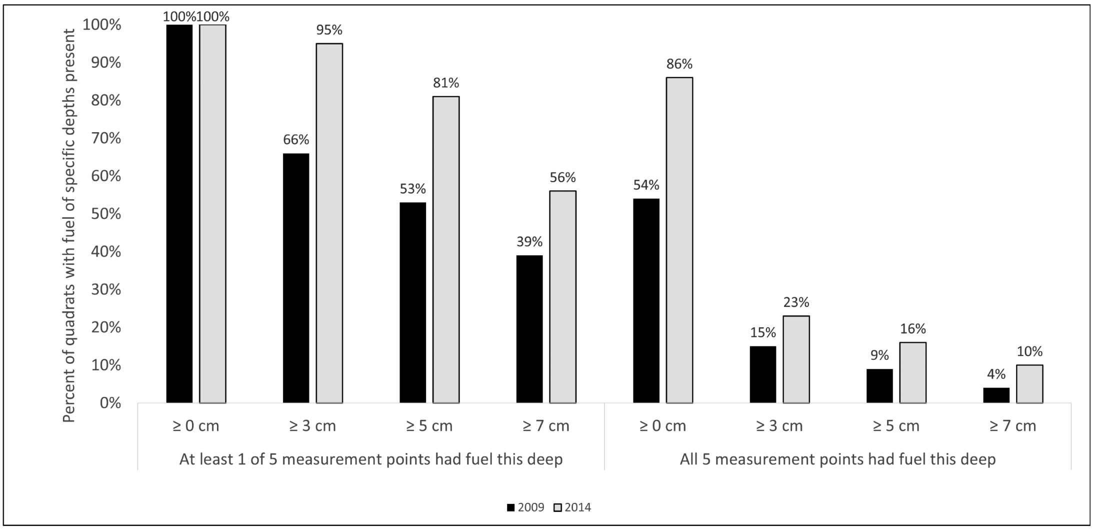

Figure 5. Percentage of quadrats with fuel of specific depths in 2009 and 2014. On the left side are the percentage of quadrats that contained at least one point of the five measured with specific fuel depths. The right group shows percentage of quadrats that had fuel at all five measurement points of the specific depths.

est shrub per quadrat reaching 21 cm (n = 120, standard error = 1.3) compared to 7 cm in height on the treated site (n = 100, standard error = 1.2). In 2014, the average tallest shrub was 114 cm on the untreated site (n = 10, standard error = 9.4) compared to 69 cm high on the treated site (n = 80, standard error = 5.6) (P = 0.004). (Figures 3c, 7).

Tree Cover and Height

Only seven planted tree seedlings coincided with vegetation quadrats on the treated site in 2014. Their average height was 141 cm, as compared to the average tallest shrub per quadrat at 69 cm on the treated site, and 114 cm on the untreated site (Figure 7). We also identified what we surmised were two naturally occurring seedlings, although the methods we used do not allow us to know for certain. These were 23 cm in height, on average. We found no tree seedlings on the untreated site in 2014.

Influence of Fuel on Shrub Growth

We found that fuel depth had an influence on both shrub cover and height by 2014. Shrub height and cover were inversely related to fuel depth. The linear regression equation for shrub cover as a function of fuel depth ($Y = 5.83 - 2.96X$, $r^2 = 0.33$, $P < 0.003$) suggests that higher fuel depth was related with lower shrub cover (Figure 8). The equation for shrub height as a function of fuel depth ($Y = 156.5 - 42.8X$, $r^2 = 0.24$, $P < 0.001$) similarly suggests that greater fuel depths were related with lower height growth of shrubs.

Discussion

Post-fire shrub dominance has been found in forests previously adapted to frequent, low-severity fires when high-severity fires occur in California (McGinnis et al. 2010, Collins and Roller 2013, Crotteau et al. 2013) and in the Tahoe basin (Russell et al. 1998, Nagel and Taylor 2005). Crotteau et al. (2013) found that shrub cover after high-severity fire was three times higher than after medium-severity burns. This increase in shrub dominance is a significant concern for regeneration of shade-intolerant trees such as ponderosa pine (Crotteau et al. 2013, Tubbesing et al. 2020) and Jeffrey pine (Nagel and Taylor 2005).

Post-fire salvage logging after high-severity fire is undertaken to both recover usable wood and to

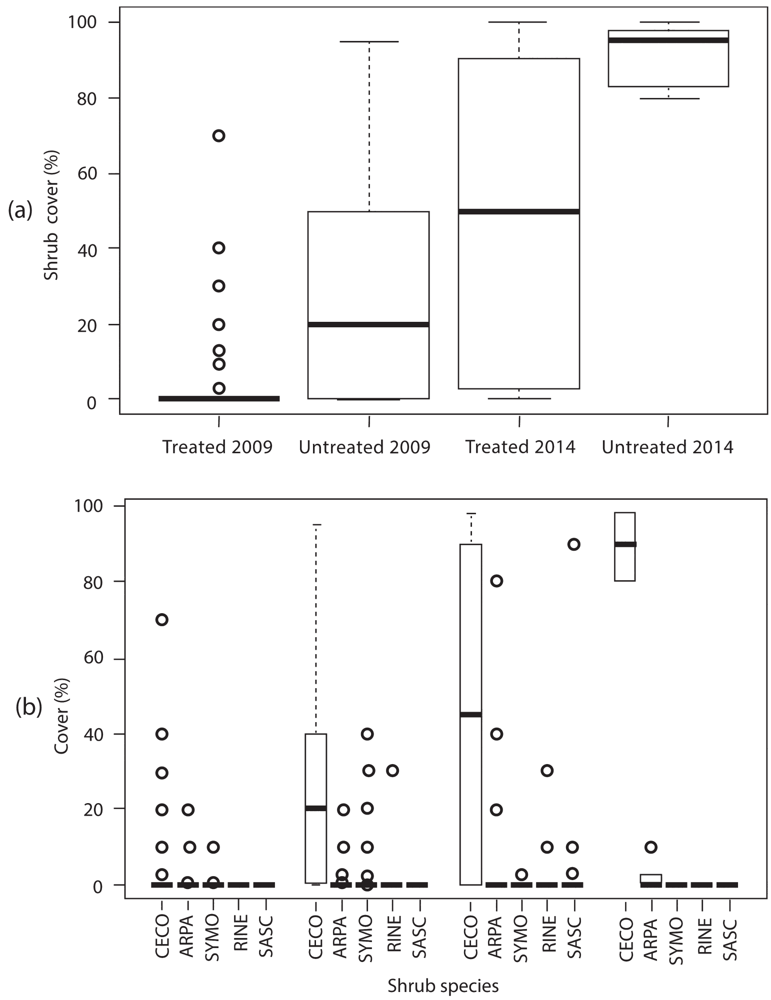

Figure 6. Shrub cover per quadrat in 2009 and 2014 on treated and untreated (control) plots. The dark horizontal line represents the median, the bottom and top box edges represent the 25th and 75th percentiles: a) cover for all shrub species combined, and b) shrub cover by species present where CECO = *Ceanothus cordulatus*, ARPA = *Arctostaphylos patula*, SYMO = *Symphoricarpos mollis*, RINE = *Ribes nevadense*, SASC = *Salix scouleriana*.

start the process of forest recovery by preparing the site for tree planting (and is often accompanied by piling and burning woody debris and spraying herbicide). Our results agree with other studies that found that salvage logging can play a part in reducing shrub dominance. Morgan et al. (2015) found salvage logging reduced canopy cover of both forbs and shrubs six years after fire in mixed-conifer forests of eastern Washington State. The same effect has been found in California frequent-fire forests (Stuart et al. 1993, Knapp and Ritchie 2016). Knapp and Ritchie (2016) found that although cover by forbs and graminoids did not differ with salvage treatment, shrub cover was significantly reduced at higher salvage intensities. However, McGinnis et al. (2010) found no significant difference between shrub cover on areas salvage logged after fire and those left untreated in the Sierra Nevada. While these studies examined the overall effect of salvage logging on vegetation, regardless of the specific mechanism causing the effect, we found that some of the reduction in post-fire shrub growth was due to the physical barrier created by widespread fuels resulting from the salvage logging and mastication on our study site. As fuel depth increased, both shrub cover and height decreased, an effect that continued over a seven-year period following the fire.

Our results are consistent with other studies that looked at the effects of mulching on understory plants to reduce erosion after wildfire, but with straw mulch rather than masticated woody debris. Dodson and Petersen (2010) found that straw mulch applied by helicopter to reduce post-fire erosion reduced understory plant cover in the second year post-fire when mulch cover exceeded 70% and when depth was at least 3 cm. Plant cover was reduced to zero when depth exceeded 7 to 8 cm, although it was rare to find areas covered by this much

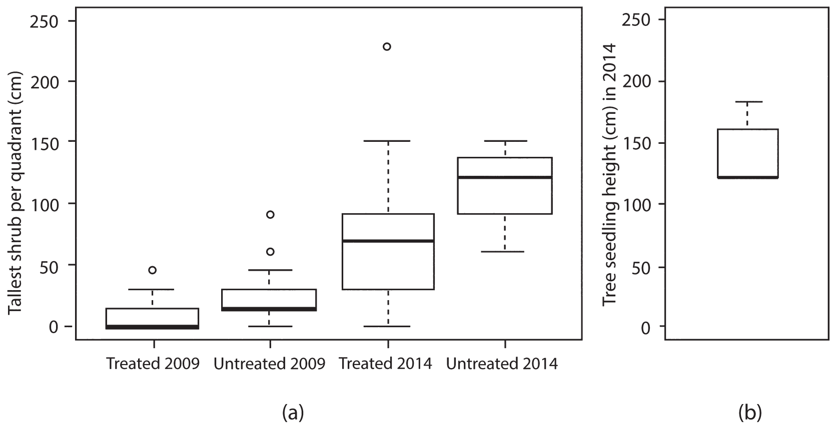

Figure 7. Height of vegetation on the treated site over time: a) tallest shrub per quadrat on the treated and untreated sites in 2009 and 2014, and b) height of planted tree seedlings found in vegetation quadrats in 2014 on the treated site (none were found on the untreated site). The dark horizontal line represents the median, the bottom and top box edges represent the 25th and 75th percentiles.

mulch. On the other hand, Bontrager et al. (2019), who investigated effects of post-fire aerial straw mulch 9 to 13 years after application, found that cover of forbs and shrubs did not differ between treated and untreated areas. However, unlike in the previous study, depth of the initial application of mulch was not measured, and very little mulch remained when they conducted their measurements. It is possible that the discrepancy in study results could be accounted for by differences in mulch application rate and extent.

Other research that investigated the effects of applying a heavy layer of wood chips to a forest understory with no fire beforehand (Miller and Seastedt 2009) may be more relevant than the straw mulch studies cited above because the type of material and application rate were similar. Researchers applied a layer of wood chips to the forest floor at the rate of 135 tonnes per hectare and an average depth of 7.5 cm, which is an application rate much like the treatment we studied. Miller and Seastedt (2009) found that the chip layer significantly lowered overall plant cover. Their study was able to determine that the effect was due to the physical effects of wood chips, rather than any potential changes of soil N availability caused by applying carbon and potentially affecting the C:N ratio. By applying woody material from the site for erosion control, the treatment we studied also eliminated the potential for introduction of new weeds that often infest straw mulch (Brontager et al. 2019).

Our results suggest that the reduced shrub cover we observed on the treated site is accounted for, in part, by the depth of fuel on site, although there are likely additional factors at play. Others have found that salvage logging itself can inhibit shrub regrowth. Knapp and Ritchie (2016) found that shrubs on their study site, dominated by mahala mat (*Ceanothus prostratus* Benth.), an obligate seeder, were suppressed in response to post-fire salvage logging. The authors speculated that salvage harvest timing played a role in suppressing shrub growth because heavy equipment was used on site, most likely crushing new post-fire shrub seedlings or resprouts that had emerged. On our study site, it is unlikely that new shrub seedlings were present because of the timing of the salvage harvest. Harvesting and mastication were completed four months after the fire, by the end of October, well before the winter rains could trigger emergence of any new shrub seedlings.

Shrub response to wildfire severity and subsequent salvage logging undoubtedly varies by regeneration mechanism. The most abundant shrubs that regrew on our study site were whitethorn and greenleaf manzanita, both of which are able to reproduce through seed germination and resprouting after fire (Hauser 2007, Reeves 2006). Both have a lignotuber that contains buds from which new stems may sprout and that stores starch to support growth. Our data suggest that new seedlings and resprouts were physically blocked from emerging in the spring by the widespread extent of woody debris on site. It is also possible that heavy equipment traversing the site during

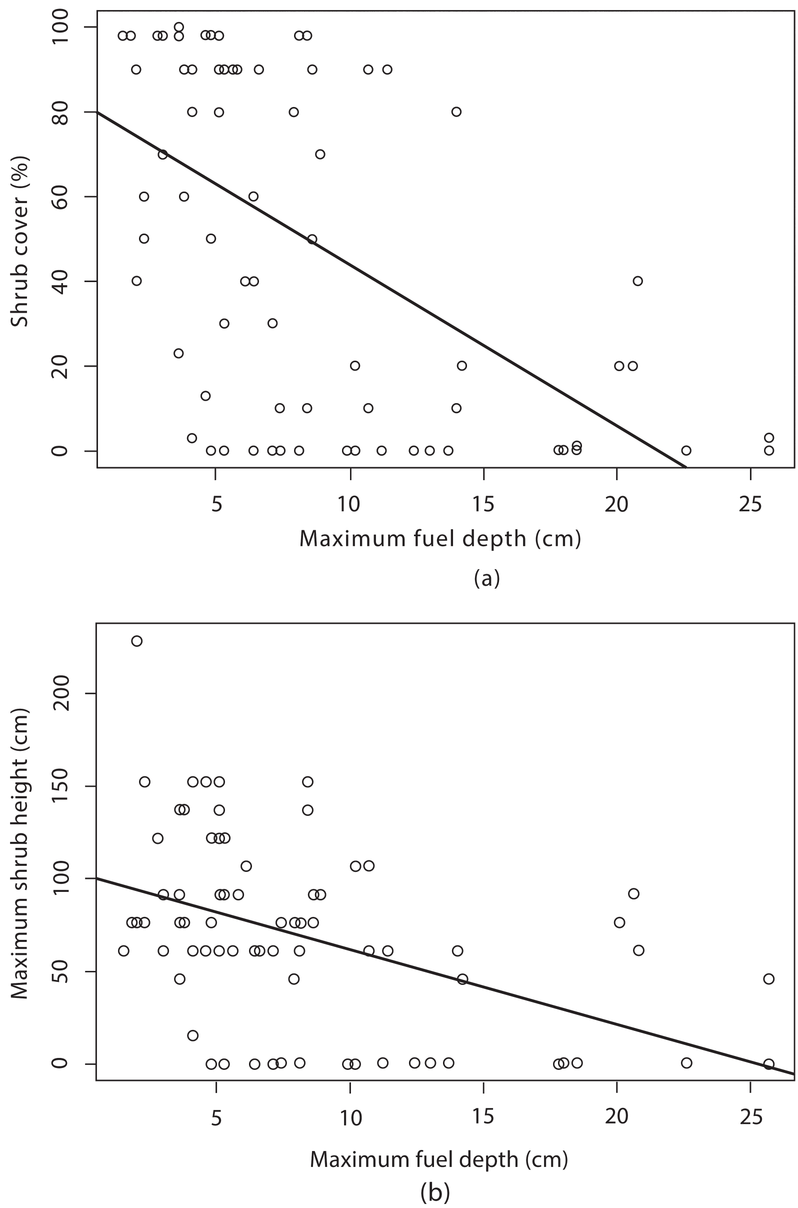

Figure 8. Regression equations showing: a) scatterplot for 2014 shrub cover as a function of maximum fuel depth ($Y = 5.83 - 2.96$ X), and b) scatterplot for 2014 maximum shrub height as a function of greatest fuel depth ($Y = 156.5 - 42.8$X).

the treatment crushed surviving underground shrub tissue and reduced the amount and vigor of their post fire re-sprouting. Additional research on mechanisms by which salvage logging activities suppress shrubs is warranted.

Control of competing vegetation, especially quickly growing shrubs, is a critical step for timely reforestation after wildfire (Crotteau et al. 2013, Stephens et al. 2020), as shrubs compete with and slow the initial growth of conifer seedlings (Nagel and Taylor 2005), especially in areas with low site quality (Zhang et al. 2006). Two past studies examined shrub and tree regrowth after a stand-replacing fire in 1882 on Angora Ridge, which is directly above our study site. Researchers found that it took from 30 to 60 years (Russell et al. 1998, Nagel and Taylor 2005) for naturally seeded trees to emerge from the shrub layer after the fire. The trees that grew into the overstory were almost exclusively shade-tolerant firs, including white fir and red fir (*Abies magnifica* A. Murray bis). Nagel and Taylor (2005) found 89% of surviving trees were fir species, while Russell et al. (1998) found that 92% of trees were fir species.

Forest managers wanting to restore burned forests and control species composition typically plant shade-intolerant tree seedlings. Our results suggest that planted trees may be given an advantage when post-fire treatments involve spreading thick woody cover across the site, which suppresses shrub growth. If rapid recovery of a forest overstory after fire is a goal, broadcasting woody residue may give an advantage to planted conifer seedlings in some forest types, especially shade-intolerant species like ponderosa pine (Crotteau et al. 2013, Tubbesing et al. 2020), and may lessen the need for other shrub control methods such as grubbing by hand, mastication, and herbicides. The ability of woody mulch to maintain soil moisture and modify temperature extremes and so promote tree growth is well established (Greenly and Rakow 1995), and could potentially provide a benefit as well, especially in the water-limited growing environment of the study area.

Although application of woody material helped reduce shrub competition, there is some concern about the fire risk associated with the extra fuel load, particularly for young tree plantations which are vulnerable to fire. Salvage logging is well known to initially increase the woody debris left on site (Donato et al. 2006, McGinnis et al. 2010, Dunn and Bailey 2015, Peterson et. al. 2015, Knapp and Ritchie 2016), and high levels of activity fuels have been shown to increase the severity of reburn in salvage-logged areas (Thompson et al. 2007).

The quantity and size of fuels in our study area differ from many post-fire logging studies, because activity fuels were masticated before being spread on site to leave a widespread and abundant layer of small-diameter masticated woody residue to reduce erosion hazard (Harrison et al. 2016). While we measured woody cover at 82% of points across the site two years after treatment, Knapp and Ritchie (2016) found only 17% combustible cover three years after treatment. This is probably because treatments in that study included transporting whole trees to landings instead of delimbing them on site, as was done on our study site. McGinnis et al. (2010) found less than 22 tonnes per ha in logged areas. In comparison, we found 190 tonnes per ha of activity fuel on site in 2008. Between 2009 and 2014, the depth of fuels actually increased at the site. By then, litter and fine fuels produced by shrubs that regrew across the site added to the depth of total fuel we measured. Activity fuel from salvage logging and mastication was also very slow to decompose at high-elevation sites where most precipitation comes during the cold winter months (Harmon et al. 1986).

The fire hazard created by the layer of masticated material left on site is difficult to assess. The amount, extent, and composition of fuels, as well as weather and topography, all have a profound effect on how an area reburns after wildfire (Lydersen et al. 2019) and salvage logging (Thompson et al. 2007). Fire behavior in masticated fuel beds varies based on the load, arrangement, moisture, and depth of fuels. Flame heights in masticated fuel beds tend to be modest, at less than 2 m, but fires smolder for longer than in natural fuels and are prone to flare up under certain conditions (Kreye et al. 2014). Fuel consumption also tends to be patchier (Glitzenstein et al. 2006). Kreye et al. (2014) reported that modeling fuel dynamics and fire behavior when overstory vegetation is present, including re-sprouting shrubs, is difficult. Certain shrubs, if allowed to regrow freely among tree seedlings, may also carry fire well (McGinnis et al. 2010).

How long fuels will persist on site is unknown at this time. Effects on shrubs were still pronounced seven years after treatment, as was the widespread abundance of surface fuels. In our high-elevation study site, dry summers and cold wet winters lead to slow decomposition of woody material (Harmon et al. 1986). In other areas, surface fuels may decompose more quickly, while dead trees that are not removed may fall more rapidly. Other researchers have found that fuel quantity on unsalvaged sites may surpass the amount on salvaged sites in as little as ten years (Knapp and Ritchie 2016) and as much as 22 years following treatment (Dunn and Bailey 2015). However, if shrub inhibition is an objective in order to promote growth of planted tree seedlings (Stephens et al. 2020), persistence of the effect is less important over time, as competition from shrubs has the greatest effect on trees in the first few years after planting (Zhang et al. 2006, Tubbesing et al. 2020). For the treatment we studied, we found that planted trees had overtopped shrubs in the first seven years on average, meaning that the competition for sunlight was mostly already decided in favor of the trees. However, inferences made about the height of planted trees in our study area should be tempered given the small number we encountered in our monitoring plots.

Managers must balance the costs and benefits of salvage logging and masticating residual material when choosing strategies to meet reforestation goals in their specific location. Our findings imply that spreading a layer of mulch created by masticating woody debris from salvage logging may inhibit shrub growth after high-severity wildfire, but this mulch layer may also pose challenges to a regenerating forest by increasing fire risk. Spreading masticated material on site may have different effects in other post-fire management contexts that have different shrub species with different regeneration mechanisms, where woody material decomposes at different rates, or where other ecological or edaphic factors affect forest regeneration. Other factors managers should consider include cost, likelihood of natural regeneration (which could be blocked by mulching), the need for shrub control (which could be reduced by mulching), and drought stress (which could be moderated by mulching), which will vary in different forested systems. Further study of the holistic effects of heavy mulching on planted tree seedlings across different forest types would provide more information for managers.

Conclusions

Salvage logging and residual woody material spread on site after a high-severity fire was shown to reduce both the cover and height of shrubs growing after the fire. Leaving a layer of woody material on site after salvage logging may reduce the amount and vigor of shrubs, giving a competitive advantage to any new tree seedlings planted and so could be a useful part of a reforestation strategy in upper elevation Sierra Nevada forests. Advantages of this approach are that use of on-site woody material eliminates the potential for introduction of weeds when mulching for erosion control. It could also be useful in areas where application of herbicide for shrub control is not desirable. This approach is quite different from the standard approach in the region which seeks to minimize activity fuels left after salvage logging by piling and burning or removal from the site. The advantages of leaving masticated material on site for shrub suppression and erosion control following salvage logging should be weighed carefully by managers against the effect of the fire hazard posed to the newly planted trees.

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