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Resilience of a ponderosa pine plantation to a backfiring operation during a mid-summer wildfire

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Abstract. The Mill Fire, which burned in north-western California during the summer of 2012, provided a unique research opportunity when firefighters implemented a backfiring operation to limit wildfire growth. This backfire was ignited and burned through research plots from a long-term study designed to determine the effects of tree density manipulation and shrub control on the growth and stand development of a ponderosa pine plantation. The objectives of this study were to examine the response of these 53-year-old trees to the backfire and to determine how the fire effects differed with plantation structure and composition. Measurements made 4 years post-fire showed that mortality rate was highly variable (from 0 to 100%) and did not relate to tree density, height of live crown, total basal area or shrub cover. Bole char height explained 65% of the variation in mortality rate. Fire appeared to spread primarily through the surface litter and killed a substantial proportion of the shrubs competing with the trees for water and nutrients. Importantly, post-fire tree growth was not significantly affected relative to pre-fire growth. A lack of negative effects of the fire on radial growth was possibly a result of release from inter-tree and shrub competition, which balanced any declines that might have been expected from bole injury or crown loss. Results from the present study demonstrate that ponderosa pine plantations could potentially be treated with managed fire (e.g. prescribed fire) without pretreatment (i.e. thinning, mastication), and still achieve good survival and improved resilience to wildfires burning under uncontrolled conditions.

Additional keywords: fire effects, overstorey density, prescribed fire, *Pinus ponderosa*, understorey shrubs, wildfire.

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Introduction

Not only have wildfires in the western USA become bigger in recent decades, but their seasons have lengthened too (Westerling *et al.* 2006; Stephens *et al.* 2014). Although many such forests have burned frequently in the past, a higher proportion of the burned area is believed to be experiencing high-severity effects today, in part owing to excess fuel accumulation (Mallek *et al.* 2013; Steel *et al.* 2015; Safford and Stevens 2017). Following high-severity disturbance, a forest might re-establish through natural recruitment, if sufficient seed-producing trees remain nearby (Welch *et al.* 2016), or be established through reforestation or active planting. However, the same trends in fire activity make it increasingly likely that forests planted following high-severity disturbances will succumb to subsequent fires well before maturity. Fire hazard both within and adjacent to plantations is often elevated because of (1) high tree densities and continuous canopies, and/or (2) the presence of understorey shrubs (Kobziar *et al.* 2009; Thompson *et al.* 2011; Zhang *et al.* 2017). High stand densities with continuous canopies, in particular, can increase vulnerability to crown fire. Understorey shrubs, which if not controlled in young stands, can become dense and produce significant biomass and higher surface fuel loads. Young trees also have thinner bark (Ryan and Reinhardt 1988) and are more vulnerable to crown scorch because they are typically of shorter stature (Regelbrugge and Conard 1993). Fire intensities that might spare larger trees can instead result in extensive mortality of smaller trees.

The value of fuel reduction treatments such as thinning and prescribed fire for reducing the severity of subsequent wildfires in historically frequent fire forests is well known (Agee and Skinner 2005; Ritchie *et al.* 2007; Safford *et al.* 2012; Prichard and Kennedy 2014). Most such studies focus on treatments applicable to mature stands (Stephens *et al.* 2013). Information on fuels treatments in young plantations is lacking, especially with prescribed fire because burning among young and relatively small trees has generally been avoided, owing to concerns about tree damage (but see Peterson *et al.* 2007; Knapp *et al.* 2011; Bellows *et al.* 2016). In California, ponderosa pine (*Pinus ponderosa* Lawson & C. Lawson) plantations cover over 162 000 ha of National Forest land (Landram 1996; J. Sherlock, pers. comm.) and ~127 860 ha of private (forest industry) lands. These numbers are expected to increase because plantations are frequently the method of choice for reforestation after stand-replacing wildfires, and both area burning in large fires (Collins *et al.* 2019) as well as high-severity patch size (Stevens *et al.* 2017) are on an upward trend.

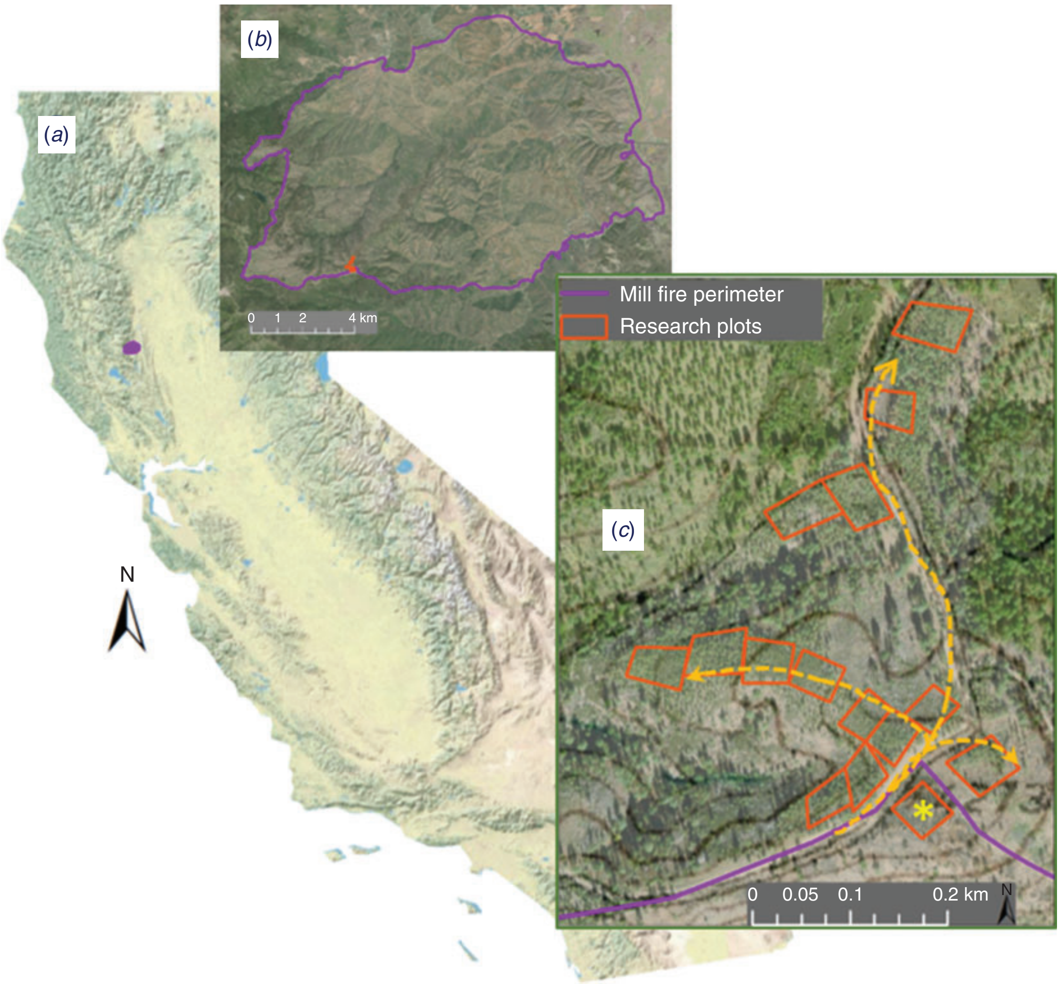

Fig. 1. (*a*) The eastside of the North Coast Ranges of California, where (*b*) The Mill Fire burned in 2012. (*c*) It included 15 research plots located in the south-west portion of the fire. All research plots except for one with a yellow asterisk were burned during a backfiring operation. The orange dash lines show the approximate progression of ignition along primary and secondary ridgetops on both sides of the road (except where the road became the ultimate control line), starting somewhere close to the most southern plot.

A backfire is an intentionally ignited fire conducted to impede the advancement of a wildfire by burning vegetation along the containment perimeter. Owing to the emergency nature of the situation, stands receiving a backfire typically are not thinned beforehand and understorey shrubs are not cut or treated with methods such as mastication, except perhaps in the immediate vicinity of the fire line. We are not aware of information about the effect of backfire on either plantation trees or natural stands, which might in part be due to the difficulty of designing and implementing such a study (Carroll *et al.* 2004). However, such information would be valuable for supporting modelling and determining resilience of plantations to fire (both wild and prescribed). If a plantation can survive a backfire with acceptable levels of mortality during dry mid-summer (July–August) conditions, this would suggest the possibility of wider use of prescribed burning to treat fuels in plantations. As opposed to wildfire, prescribed burns are typically conducted on the shoulders of the wildfire season when temperatures are cooler and fuels somewhat moister, potentially moderating the effects on trees (Harrington 1987, 1993). Fire management operations during the Mill Fire in July 2012 created an opportunity to study impacts of a backfire (Fig. 1). Firefighters chose this area for a backfire because it was located on a mountain ridge road just inside the Mill Fire control line and ultimate fire perimeter. The area just so happened to run through an existing study. In the original study design, experimental density plots of planted ponderosa pine were established on both sides of the road, with three levels of understorey removal treatments nested within five tree density treatments, each replicated three times. Trees were last measured in 2005, 46 years after planting.

The goals of the present study were to quantify the resilience of a ponderosa pine plantation to a backfiring operation, and to glean information that might help to broaden options available to managers concerned about managing fuels in plantations to improve resilience to future fires. We addressed four specific questions: (1) how did the backfire operation affect tree survival; (2) what influence did plantation density, measured by trees per hectare (TPH), and understorey shrub cover have on the effect; (3) was there an interaction between tree density and understorey shrub cover and how did this affect the survival of plantation trees; and (4) was there any effect of fire on growth of surviving trees?

Methods

Study site and background

Following a wildfire in August 1959 in Mendocino National Forest, Northern Coast Range of California, the majority of the burned area was salvaged, with remaining woody material windrowed and the site disked (Oliver 1984). In 1960, 2-year-old pine seedlings were planted at densities ranging from 1422 to 3047 TPH. The soil is a Dystric Lithic Xerochrept (Maymen series) derived from Pre-Cretaceous metasedimentary rock, is well drained and has a low water-holding capacity; it is therefore a relatively poor site for tree growth (Oliver 1984). Depth to lithic contact varies from 20 to 33 cm.

The 15 study plots (39.2804′N, 122.6710′W, elevation 1300 m) were installed within 15 ha of the plantation in 1970 to assess the density effect on future stand growth and development. At that time, trees averaged 2.8 cm in diameter at breast height (1.37 m; DBH) and 1.8 m in height. At the start of the experiment, the original stand was thinned from below, targeting four spacings of 2.1 × 2.1 m, 2.4 × 2.4 m, 3.0 × 3.0 m and 4.3 × 4.3 m, with three replicates per spacing in a randomised block design. Three unthinned control plots were initially included, although the stand density was similar to the 2.1-m spacing treatment and subsequently both un-thinned plots and 2.1-m spacing plots were treated as controls, resulting in a total of six control plots. The final tree densities achieved were ~2200 TPH for both the unthinned and 2.1-m spacing (control), 1680 TPH for the 2.4-m spacing, 1080 TPH for the 3.0-m spacing and 550 TPH for the 4.3-m spacing. All trees were tagged at plot establishment and were measured approximately every 5 years from 1970 to 2005 (Oliver 1984; Zhang *et al.* 2006). Measurements included DBH, height (HT) and height to live crown base (HLC).

Within 15 years of planting, shrubs – predominantly hoary manzanita (*Arctostaphylos canescens* subsp. *sonomensis* Eastw.), a non-resprouting obligate seeding species – had established in abundance within the plantation, forming a relatively uniform understorey. In 1975, a shrub removal treatment was superimposed on each stand density plot. Each 0.10-ha plot was divided into three equal subplots and three shrub removal treatments were randomly assigned: (1) all shrub tops were manually removed (V0); (2) every other shrub top was removed (V0.5), leaving ~15% residual shrub cover; and (3) shrubs were untouched (control) (V1), with ~35% shrub cover at time of treatment. Shrub stem removal treatments were only conducted once, with the exception of a few scattered individuals of resprouting species. While seed is generally abundant in the soil in such forests (Knapp *et al.* 2012), germination in many *Arctostaphylos* species is thought to be stimulated in part by chemicals in char, and little germination typically occurs beyond the first year after fire (Keeley 1991). In the following decades, although uncut shrubs continued to grow, number of shrub stems remained relatively unchanged (W. Oliver and J. Zhang, pers. obs.). During measurements in 1991, the average shrub crown cover had increased to 90% in V1 subplots, 47% in the V0.5 subplots and remained at or near zero in the V0 subplots.

On 7–18 July 2012, the Mill Fire occurred within the Mendocino National Forest, Grindstone Ranger District (Fig. 1). The fire started ~15-km south-west of the Colusa County community of Stonyford and burned 11 939 ha. The research plots were burned when the backfire was ignited along and interior of the south-west control line and ultimate perimeter of the Mill Fire (Fig. 1). The backfire was ignited with drip torches from high points along primary and secondary ridges and slowly guided to lower elevations. It was initiated sometime during the night of 12 July and completed in the afternoon of 13 July (Mill Fire Incident Status Summary (ICS-209), 2012 Mill Fire Incident Action Plans; J. Tishner, Fire Management Officer, Grindstone Ranger District, Mendocino National Forest, pers. comm.). Given the location of the plots on or close to the ridge top, most if not all plots likely burned during the initial phase of the burnout. Although it was hot and dry during the day (peak temperature = 38°C, with relative humidity (RH) of 11%), temperature was much lower and RH much higher overnight, dropping to 16°C and 60% in the early morning hours (Fig. 2*a*). Winds overnight were relatively calm and remote automatic weather station fine dead fuel moisture was estimated to range from 4 to 5.5% during the time of the backfiring operation (Fig. 2*b*). The backfire burned through 14 of the original 15 research plots. The one unburned plot (Fig. 1), a density control in the original study, was excluded from further analysis.

The 2016 post-fire measurements

In 2016, 4 years after the Mill Fire, we re-measured the same tree variables as previously collected, recorded fire-caused mortality and measured bole char height as the height to the highest blackened bark on the bole for all living trees. Shrub aboveground biomass (living and dead) was measured by establishing 1 × 1-m biomass subplots in the centre of each subplot. Shrubs grown and rooted within the subplot boundary were harvested and weighed in the field, and separated into living and dead biomass. A subsample of each, including stems, branches and leaves was brought back to the laboratory, weighed and dried in an oven at 80°C for at least 48 h until a constant mass was reached. We recorded the dry weight and extrapolated subsample dry weight back to the whole-sample basis.

We collected one increment core at breast height (1.37 m) from six living trees in each subplot, with the exclusion of two subplots that had almost 100% mortality. Cored trees were randomly selected stems stratified by small, medium and large DBH classes, with the DBH class sizes allowed to vary so as to best represent the stem diameter range in each individual subplot. In the laboratory, the increment cores were dried, mounted, sanded and visually cross-dated to account for missing rings using standard dendrochronological methods. Ring widths were then measured to 0.001 mm accuracy using a scanned image (1200 dpi) in CooRecorder, version 8.1.1 (Larsson 2015).

To calculate the climatic moisture index (CMI; Daly *et al.* 2008), we obtained site-specific monthly weather data for precipitation and average temperature from the PRISM

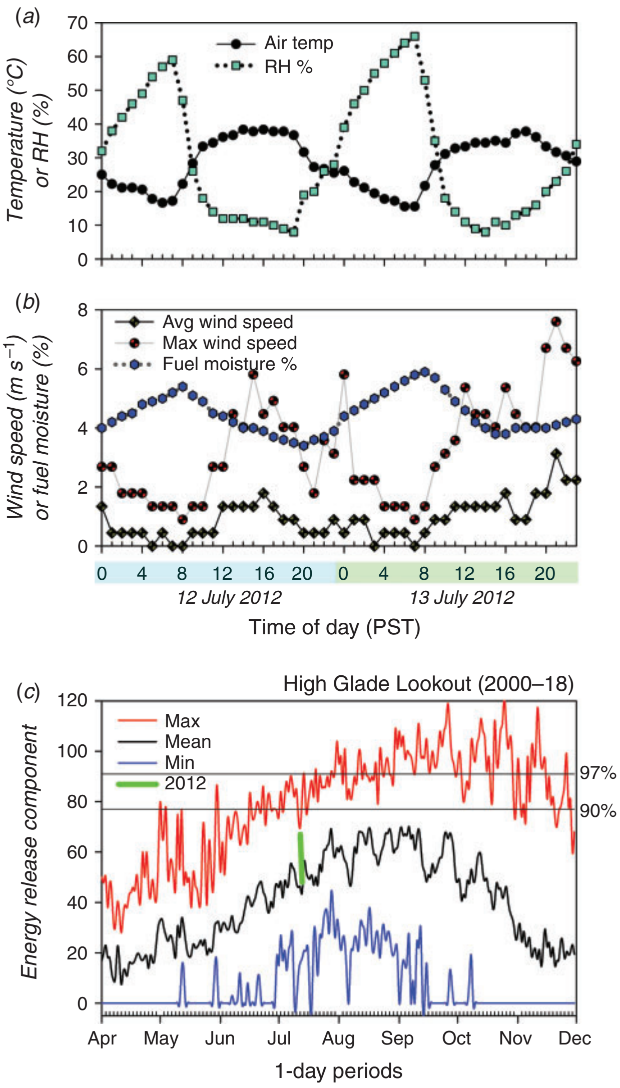

Fig. 2. (*a*) Hourly air temperature (°C) and relative humidity (RH, %), (*b*) moisture content (%), average wind speed (m s^{-1}), and maximum gust wind speed (m s^{-1}), which were recorded with a portable remote automatic weather station (RAWS) during the times (PST, Pacific Standard Time) when the backfire was conducted. Plots were burned between the night of 12 July, when backfiring in the area began, and the afternoon of 13 July. (*c*) Average daily maximum, mean and minimum energy release component values were calculated from the High Glade Lookout RAWS station (data from 2000 to 2018), for the period from 1 April to 30 November. The range of average daily values for 12–13 July 2012, during the Mill Fire backfiring operation, are shown with a vertical green line.

(Parameter-elevation Regressions on Independent Slopes Model) climate group at Oregon State University. Daily estimates of energy release component (ERC, a measure of the amount of available energy per unit area at the head of the fire, and a function of live and dead fuel moistures) were calculated with Fire Family Plus 4.2 (https://www.firelab.org/project/firefamilyplus), using weather data for the years 2000–18 from High Glade Lookout (National Weather Service station ID 41402, elevation 1475 m), located 14.3 km south-west of the study site.

Data analysis

From the tree measurements, we calculated subplot-level basal area (BA), TPH, quadratic mean diameter (QMD), HT and HLC. All original measurement variables were analysed based on a split-plot, randomised, complete-block with a repeated-measures design. Tree density was the main plot effect, shrub cover the subplot effect and year the repeated-measure (sub-subplot) effect. All were set as fixed effects while block was set as a random effect in SAS PROC MIXED (SAS Institute Inc., Cary, NC, USA). The full statistical model is:

$$y_{ijkl} = \mu + \alpha_i + \varepsilon_{1ik} + \beta_j + \alpha\beta_{ij} + \gamma_k + \varepsilon_{2ijk} + \phi_l + \alpha\phi_{il} + \beta\phi_{jl} + \alpha\beta\phi_{ijl} + \varepsilon_{3ijkl}$$

where y_{ijkl} is the dependent variable summarised for the ith tree density, jth shrub cover, the kth block and lth year; μ is the overall mean; α_i and β_j are the fixed effects of the ith tree density ($i = 1, 2, 3$ and 4) or jth shrub cover ($j = 1, 2$ and 3); φ_l is the fixed effect of the lth year ($l = 1, 2, \ldots 7$); γ_k is the random effect of the kth block ($k = 1, 2$ and 3), $\gamma_k \sim N(0, \sigma_B^2)$; ε_{1ik} is an experimental error to test the main plot effect; ε_{2ijk} is an experimental error to test the subplot effect; and ε_{3ijkl} is for the rest of the terms, $\varepsilon_{1ik} \sim iid\, N(0, \sigma_{e1}^2)$, $\varepsilon_{2ijk} \sim iid\, N(0, \sigma_{e2}^2)$ and $\varepsilon_{3ijkl} \sim iid\, N(0, \sigma_{e3}^2)$.

For each variable, residuals were examined to ensure that statistical assumptions of normality and homoscedasticity were met. When they were not, a natural log-transformation was applied. During the model selection process, we selected the model with the minimum Akaike Information Criterion. Multiple comparisons among treatments were conducted for least-squares means using the Tukey–Kramer test by controlling for the overall $\alpha = 0.05$.

For tree ring data, we only used the mean raw widths for the years 2009 through 2015 to estimate the contrasts between pre- and post-fire for a duration of 1 (2011 *v.* 2013), 2 (2010–11 *v.* 2013–14), and 3 (2009–11 *v.* 2013–15) years with PROC MIXED. To eliminate any influence of drought during recent years, we used CMI – the ratio of precipitation and potential evapotranspiration – as a covariate to account for yearly trends and other treatment effects on tree-ring width. Relationships were analysed with post-fire mortality rate as the dependent variable and bole char height, plus previously measured BA, QMD, HLC and TPH as independent variables.

Results and discussion

Treatment effect on plantation dynamics

Across all measurement years since 1975, tree density had a significant positive effect on BA, but no effect was detected for total height or HLC (Table 1, Table S1). Although QMD was 20% less at the highest tree density compared with the lowest tree density in 2005, the effect of density was not statistically significant ($P = 0.065$). All variables except for HLC differed significantly among the three understorey shrub manipulation treatments. As expected, the repeated-measure effect (year) and year-by-shrub treatment interactions were highly significant for all variables. With the exception of the year-by-density interaction on BA ($P < 0.001$), no interactions (including the year-by-density and the year-by-shrub-by-density) were significant for any other variables (Table 1).

Table 1. A summary of analyses of variance for quadratic mean diameter (QMD), height (HT), height to live crown (HLC), basal area (BA), bole char height and post-fire shrub fuel load with a mixed model for a ponderosa pine plantation in northern California
Bold values reflect level of significance ($\alpha = 0.05$)

Source of variation	Number d.f.	Denominator d.f.	QMD (cm)	HT (m)	HLC (m)	BA ($m^2 ha^{-1}$)	Bole char height (m)	Shrub biomass ($Mg ha^{-1}$)
Tree density (D)	3	6	0.065	0.241	0.117	**0.003**	0.406	0.406
Shrub	2	16	**<0.001**	**<0.001**	0.755	**<0.001**	**<0.001**	**<0.001**
D × Shrub	6	16	0.450	0.400	0.186	0.278	**0.005**	0.005
Year	6	143	**<0.001**	**<0.001**	**<0.001**	**<0.001**		
Year × D	18	143	0.570	0.070	0.620	**<0.001**		
Year × Shrub	12	143	**<0.001**	**<0.001**	**<0.001**	**<0.001**		
Year × D × Shrub	36	143	0.913	1.000	0.993	0.995		

Reducing stand density to enhance growth of residual trees has been a standard forest practice for many years. Therefore, that an effect of density on QMD (Fig. 3) was not stronger is somewhat surprising, because previous studies for ponderosa pine have reported substantially larger trees in lower density plots compared with higher density plots (Zhang *et al.* 2006, 2013*b*). However, it may take a longer time for differences in QMD and height to develop on a lower quality site, because of slower stand growth (Zhang *et al.* 2016). Trees on this site are growing in extremely shallow soil (<25 cm) and are also likely competing with shrubs for moisture and nutrients (Zhang *et al.* 2006, 2013*a*).

A strong positive influence of tree density on BA was consistent with previous density studies on ponderosa pine (Zhang *et al.* 2013*b*); the higher density plots will carry more BA than lower density plots until the onset of self-thinning or mortality caused by a disturbance event. In the present study, the relationship between density and BA was clearly demonstrated before the fire in the V0 treatment (Fig. 3). The trends varied for the two intermediate tree densities in the V0.5 and V1 treatments. After the fire, the highest density plots suffered from heavier mortality and ended up carrying a similar amount of BA as the two intermediate density plots.

Because tree height and stem diameter often dictate whether young trees survive a fire (Ryan and Reinhardt 1988; Regelbrugge and Conard 1993; Knapp *et al.* 2011; Hood *et al.* 2018), factors that impede tree growth might affect resilience to wildfire until sufficient trees in the stand reach a more fire resistant size. In the present study, BA of trees in 2005 was 124% and 70% greater in V0 (full shrub removal) and V0.5 (half shrub removal) treatments compared with V1 (no shrub removal), which mirrors findings of other studies of shrub competition in young ponderosa pine plantations (McDonald and Fiddler 2010; Zhang *et al.* 2013*a*). Tree growth rate can decrease when ground cover of woody shrubs approaches as little as 20% (Shainsky and Radosevich 1986), with the effect persisting well after overstorey canopy closes (Zhang *et al.* 2006, 2013*a*). The long duration since shrub stems were cut might explain why the impact of the V0.5 and V1 treatments on tree size and height growth was similar and nonsignificant at most tree densities (Fig. 3). In fact, the post-fire total and live shrub biomass measured in 2016 only showed a significant difference between V0.5 and V1 in the lowest tree density treatment (550 TPH). Plenty of time had transpired for existing stems to expand their branches into any available space (Zhang *et al.* 2006). However, post-fire measurements of shrub biomass might also be less accurate because the amount of biomass consumed by fire could not be determined (Table 1; Fig. 4).

Fire-caused tree mortality

Across treatments, tree mortality owing to the 2012 Mill Fire averaged 24%, equivalent to an average BA loss of 2.5 $m^2 ha^{-1}$, or 14% of the 2005 BA, suggesting that mortality was concentrated in the smaller diameter classes. We did not find any significant relationship between mortality and pre-fire HLC, QMD and BA (r^2 <0.14, $P > 0.10$; Fig. 5). Our expectation was that the plots with higher density and smaller trees with lower crowns would have suffered greater mortality. While the trees were planted more than 50 years prior, they were still quite small for their age, averaging only 12.0 cm in DBH in 2005. Live trees that survived the fire averaged 14.4 cm in diameter in 2016. While we did find that the subplots with the highest mortality rates (>20%) were mainly in the high density treatments (1680 and 2200 TPH), other factors appeared to overwhelm the influence of tree density and size.

Shrub biomass in V1 and V0.5 plots was substantial (>50 Mg ha^{-1} to more than 300 Mg ha^{-1} of non-consumed shrub biomass post fire, Fig. 4), and because dense montane chaparral shrubs often burn at stand-replacing intensity with high flame lengths (van Wagtendonk and Botti 1984; Scott and Burgan 2005), we expected higher tree mortality in the treatments with the highest shrub cover. However, no relationship between shrub treatment and tree mortality was found (Fig. 5). Instead, the 12 subplots where overstorey tree mortality rate exceeded 20% covered the gamut of shrub treatment – five V1 (no removal) subplots, three V0.5 (half removal) subplots and four V0 (full removal) subplots. This suggests that shrubs may not have been an ‘effective’ fuel under the weather and moisture conditions during the backfiring operation. In fact, shrubs may have acted, either directly or indirectly, to suppress fire behaviour. Bole char height differed significantly among shrub treatments ($P < 0.001$) but not among tree densities (Table 1), and was the lowest in the V1 (no removal) treatment and highest in the V0 (full removal) treatment (Fig. 6), suggesting that lower flame lengths occurred in plots with the most shrubs. Bole char height, although only roughly and imperfectly related to flame length and fireline intensity (Alexander and Cruz 2012), explained over 65% variation of mortality rate in the present study (Fig. 7). Numerous studies have shown a similar positive association between fire damage measures (bole char height, scorch and torch height, scorch and torch percentage) and tree mortality in ponderosa pine or mixed conifer stands (Knapp *et al.* 2011; Safford *et al.* 2012).

Rather than varying with tree density and shrub removal treatments, variation in mortality was likely more strongly influenced by other factors, such as ignition pattern, wind direction in relation to aspect and direction of fire spread, or fuel manipulations that preceded the backfire. For example, some areas of higher mortality (including one of the plots that suffered nearly 100% tree mortality) appeared to be in strips perpendicular to the slope, suggesting an association with a strip head fire during ignition. The other plot with nearly 100% tree mortality was adjacent to a pile of bulldozed shrubs. To understand why treatments did not influence mortality, it is also instructive to look at the weather and fuel conditions under which the plots burned, as well as interactions among stand dynamics, fuels and potential fire behaviour in greater depth. Although montane chaparral shrubs will burn vigorously and at stand replacement intensity under extreme conditions, the conditions that allow fire to spread are narrower. Possibly because of its drought-resistant nature, manzanita has a relatively high water content and a high ignition temperature for both stems and leaves (Engstrom *et al.* 2004). Surface fuels in the understorey are generally sparse, and either or both wind and slope alignment

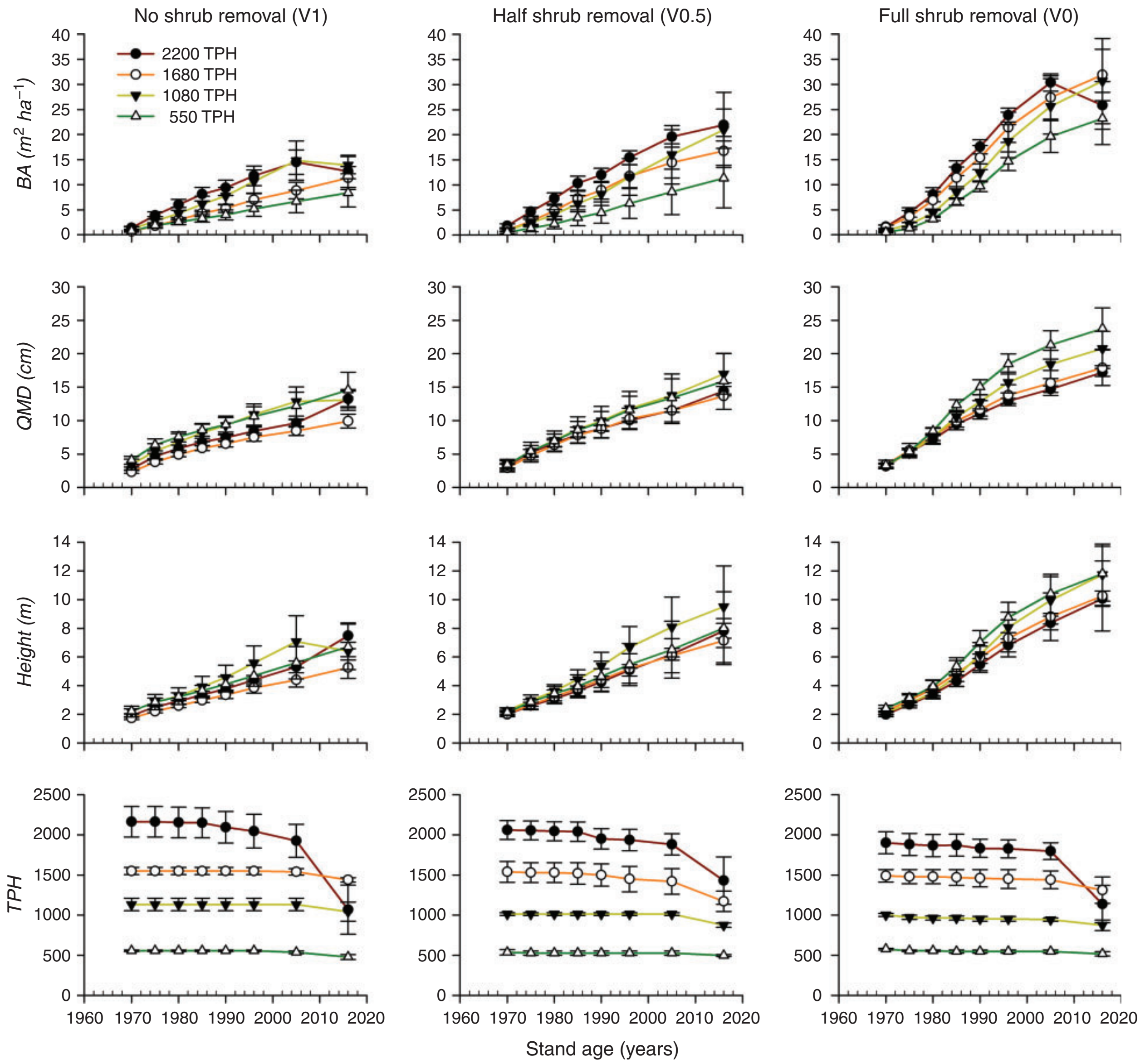

Fig. 3. Means and standard errors of basal area (BA), quadratic mean diameter (QMD), height and trees per hectare (TPH) for ponderosa pine grown in different densities under understorey shrubs naturally developed (V1), half removed (V0.5) and completely removed (V0) on the eastside of the North Coast Ranges of California.

with the direction of fire spread (combined with low live fuel moistures) are therefore often required to propagate fire through the crowns of live shrubs. Nagel and Taylor (2005) and Airey Lauvaux *et al.* (2016) both reported a fire return interval in dense montane shrubs about double that found in adjacent forest, possibly as a result of not only fuel accumulation differences but differences in conditions suitable for burning. The ERC

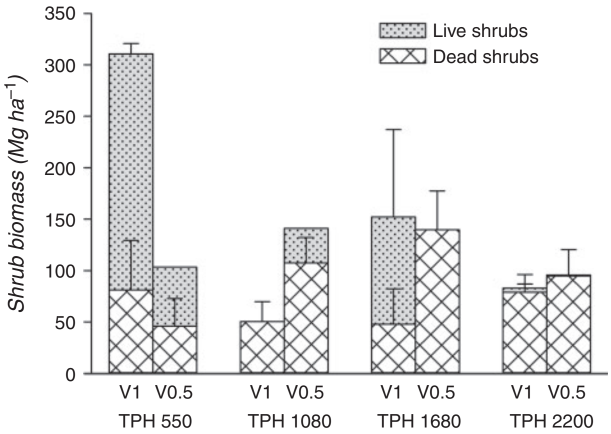

Fig. 4. Post-fire shrub biomass in pine plantations of four densities (TPH, trees per ha) measured in 2016 under treatments of no shrub removal (V1) and half shrub removal (V0.5). Few if any shrubs are found in the V0 (full removal) treatment, and data were not collected. Most of the standing dead biomass was the result of mortality caused by the 2012 Mill Fire.

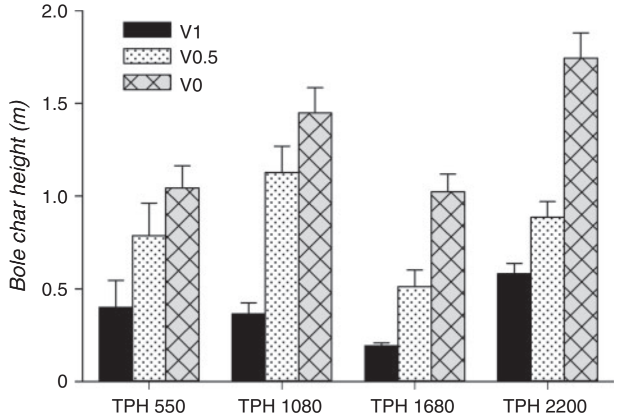

Fig. 6. Means and standard errors of bole char height measured in 2016 on trees grown on density-by-shrub treatment plots. V1, no shrub removal; V0.5, half shrub removal; V0, all shrub removal; TPH, trees per ha.

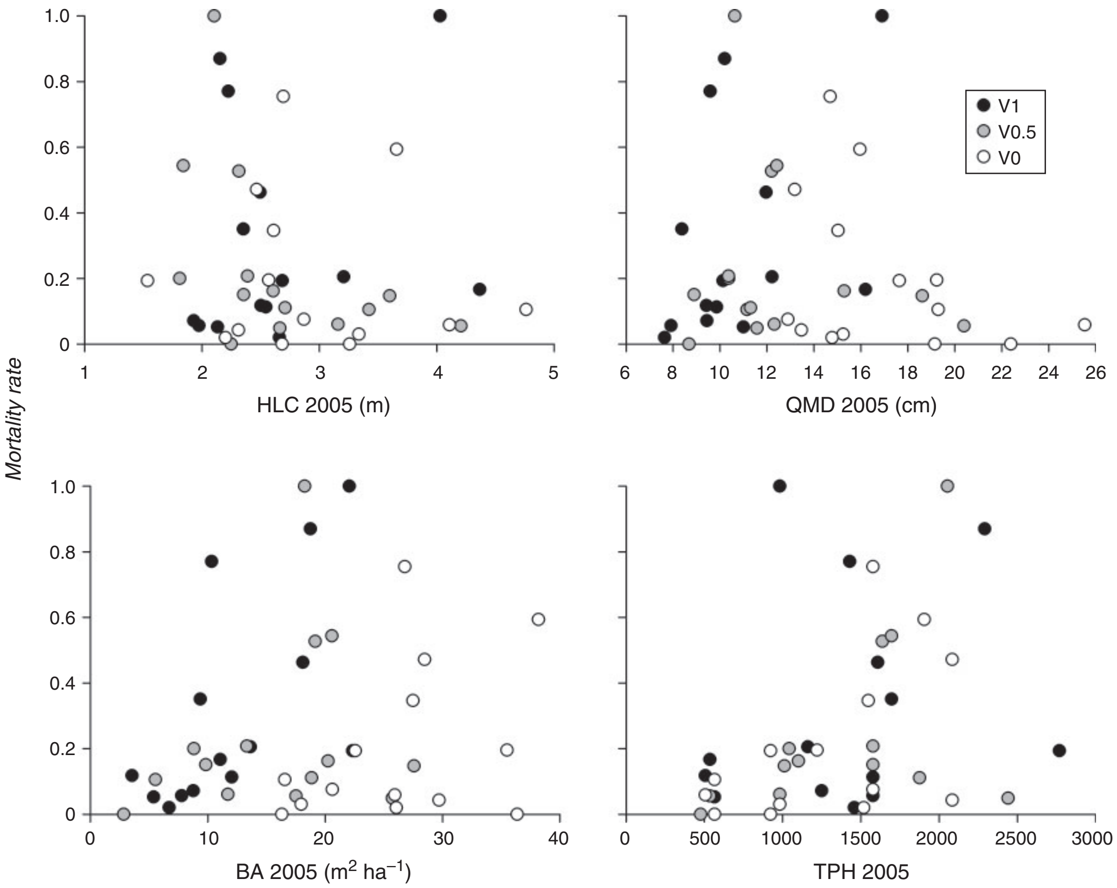

Fig. 5. Fire-caused mortality relationships with height to live crown (HLC), quadratic mean diameter (QMD), basal area (BA) and trees per hectare (TPH) from the previous (2005) measurements for all plots. V1, no shrub removal; V0.5, half shrub removal; V0, all shrub removal.

values were calculated as 67 and 48 on 12 and 13 July 2012,the day(s) the study area was impacted by the Mill Fire (Fig. 2c). Although above average for the date, they were both at less than 90% percentile in ERC for the season. In addition, live and dead fuel moistures are typically not yet at peak dryness in mid-July. Peak ERC values frequently occur somewhat later in the fire season – often August or September (Fig. 2c). These plots also

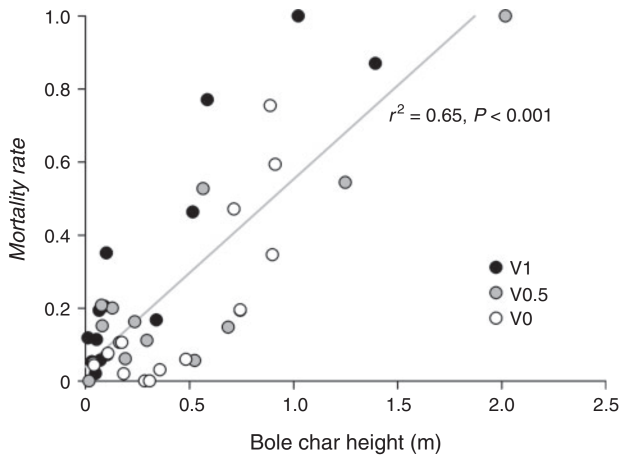

Fig. 7. Relationship between mortality rate and mean bole char height for 43 subplots. V1, no shrub removal; V0.5, half shrub removal; V0, all shrub removal.

likely burned at night, under conditions that would moderate fire behaviour. Our initial post-fire field visit (2012) suggested that even in stands with heavy shrub cover, fire spread primarily on surface litter, not through shrub crowns (Fig. 8). The result was a top-killing of a substantial number of manzanita stems.

Another explanation for lack of a relationship between shrub cover and tree mortality might relate to the needle fall, which comprises the majority of surface fuels and possibly interacted with shrubs to influence fire behaviour in these stands (Agee and Skinner 2005; Mitchell *et al.* 2009). Rate of litter and fine woody fuel accumulation has been shown to be positively correlated with stand BA (Lydersen *et al.* 2015), and a Ritchie *et al.* (2013) model for ponderosa pine in north-eastern California indicates a positive relationship between tree density and rates of litter fall, suggesting more surface litter was present in the highest BA and/or high density plots. Both bole char height (Fig. 6) and tree mortality (Fig. 3) were numerically greatest in plots with the highest BA and highest tree density, which is consistent with this hypothesis. Given that the fire largely did not involve the shrubs, litter and surface fuel differences might also explain why bole char height was greatest in the full shrub removal plots. Shrub control allowed trees to grow faster and increased BA at the plot level, likely meaning more needle litter and surface fuel were present. Greater bole char height (and therefore greater fire intensity) in the full shrub removal treatment did not, however, result in higher tree mortality, because trees were larger and taller, and therefore more resistant to fire.

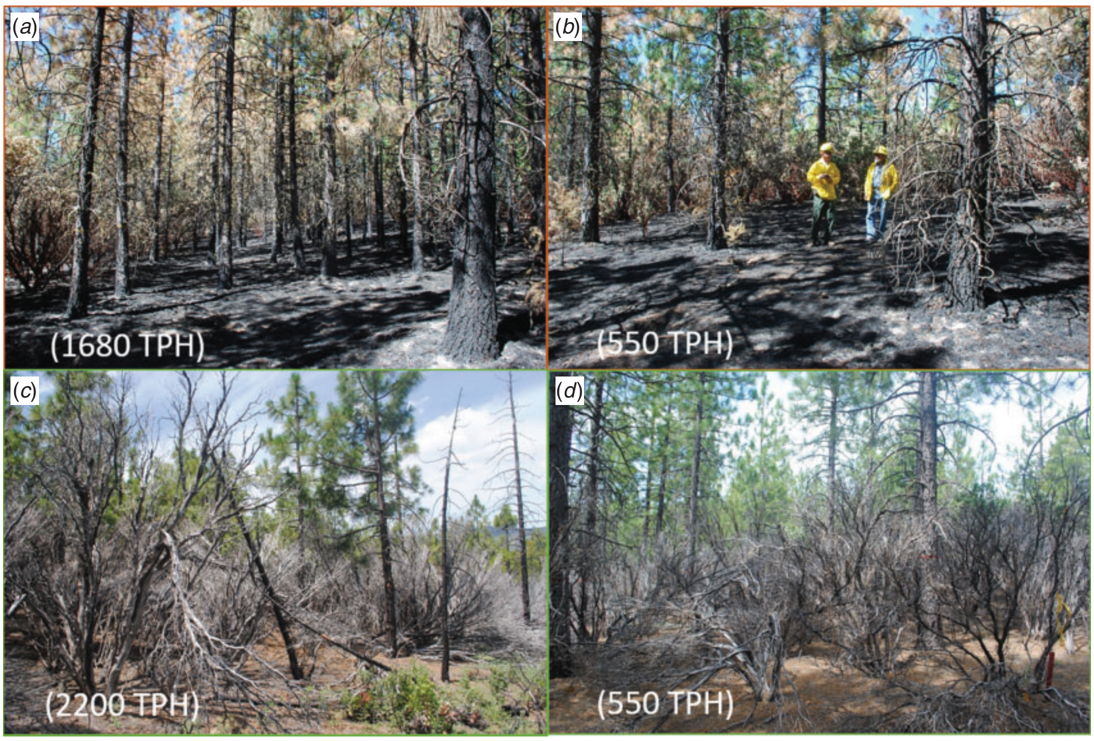

Fig. 8. Two full shrub removal subplots with tree densities of (*a*) 1680 and (*b*) 550 per hectare, shortly after the 2012 Mill Fire. The bottom two photos, taken in July 2019, show brush and trees in two subplots where shrubs were not treated before the Mill Fire, at tree densities of (*c*) 2200 and (*d*) 550 per hectare.

Fire impact on growth of live trees

For trees that survived the fire, we expected growth declines at least in the short-term as a result of the effects of bole char, crown scorch and fine root damage. We initially found that all contrasts among 1, 2 and 3 years pre- and post-fire were significant ($P < 0.001$), with a higher growth rate before the fire than after the fire (Fig. 9). Trees seemed to recover more quickly on the higher density plots than on the lower density plots, and in the no shrub-controlled treatments (V1) than in the shrub-controlled treatment (V0). However, the fire also occurred at the start of a drought and the CMI showed the same yearly trend as the tree-ring width, with the exception of 2013. When used as a covariate for further ring-width analysis, CMI was significant ($P < 0.001$) and none of the density-related terms was significant ($P > 0.502$). Therefore, we reran the reduced model by eliminating density-related terms and also compared pre- and post-fire contrasts by shrub manipulation. All three contrasts among vegetation treatments were not significant ($P > 0.171$). Therefore, after accounting for climate, we could not detect any influence of the backfire on post-fire tree growth. This was somewhat surprising, given observations immediately post-fire of considerable crown scorch in several plots. Reduced growth has been reported when the percentage of crown volume scorch exceeded 50% (Lynch 1959), but lower levels of scorch have been shown to result in increases in height and diameter growth (Morris and Mowat 1958). Landsberg *et al.* (1984) found that periodic annual growth increments in a ponderosa pine stand were reduced for trees still alive four growing seasons after prescribed fire in Central Oregon, with the magnitude of reduction related to the level of fuel consumption. Increased fuel consumption caused greater rates of reduction in tree growth. In the present study, if growth of surviving trees was reduced by fire damage, it might have been balanced by growth increases resulting from reduced shrub and interspecific tree competition.

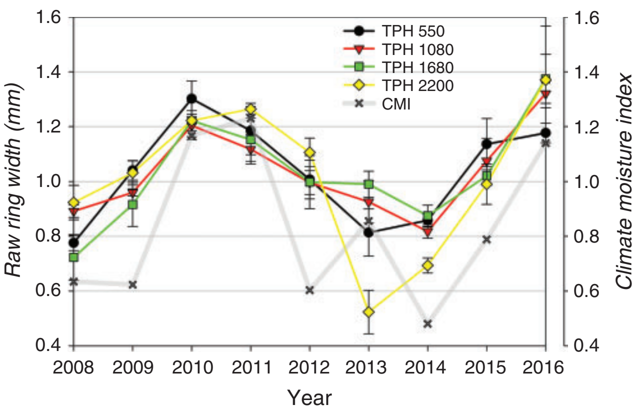

Fig. 9. Average raw ring width from at least 18 tree cores collected from large, intermediate and small DBH (diameter at breast height) trees from different combinations of tree density and shrub control in plots in northern California. The Mill Fire occurred in 2012. Climate moisture index (CMI) is the ratio of annual estimates of evapotranspiration over precipitation for the period from the previous October to the end of September in the current year. TPH, trees per ha.

Management implications

Backfires or burnouts (burning fuels between a control line and fire edge) during suppression operations can sometimes lead to high-intensity fire and greater-than-desired severity if firing must be done quickly (or otherwise if not conducted with resource objectives being a primary focus; Ingalsbee 2015). However, the results of this backfire demonstrate that positive outcomes, including greater forest resilience to subsequent wildfire, can also occur under certain burning conditions, even in a forest with a structure vulnerable to high-severity effects. The lack of substantial negative impact on tree mortality and annual growth suggest two possibilities. First, burning to improve resilience of plantations to wildfire might be possible under a greater range of conditions than is currently used in seasonally dry coniferous forest of the western USA today. Prescribed burning in such forests is typically done during narrow windows in the autumn and spring, at the margins of the fire season (Knapp *et al.* 2009). If a plantation consisting of relatively small trees can survive a backfire under mid-summer conditions, other options for using fire might be available. With studies having shown crown scorch to be central to tree survival with fire in young stands (Knapp *et al.* 2011), taking advantage of burn windows when temperatures are cool and RH high (including at night) and using ignition patterns that minimise crown scorch might provide expanded opportunities for prescribed fire. Crown scorch is more of an issue during the day when temperatures are hot, because needles require far less heating to reach critical temperatures that result in tissue death (Van Wagner 1973). Second, pre-treating shrub fuels in plantations to avoid undesirable fire behaviour and effects might not always be necessary before burning. Some common shrub fuels treatments include mastication or hand removal, both of which are time consuming and expensive (Jain *et al.* 2012). With the appropriate burning conditions, such as those characterising this backfiring operation, surface fuels and future shrub competition can be substantially reduced without causing excessive tree mortality or tree injury and loss of future growth. Successfully reducing shrubs in this way might depend on burning when the surface litter is quite dry. When moist, the light and patchy surface litter typically found under shrubs does not carry fire very well. In addition, if release from shrub competition is the management goal, findings from the present study suggest that fire might, under certain conditions, be used as an alternative to herbicides or hand removal for control of some non-resprouting species.

While the outcomes from the backfiring, including reduced surface fuels and thinning some stands accompanied by increased growth of residual trees, were largely beneficial, additional treatments will be necessary to maintain stand resilience. For one, most of the density treatments still have far too many trees. Based on contemporary plantation management protocols to increase productivity and health for sites of similar quality, a 53-year-old stand should have been thinned multiple times to fewer than 250 residual trees per hectare. Our plots all carried at least double this tree density before the fire (age 53) (Fig. 5), and most remained above this threshold after the fire. Mechanical thinning might be necessary to move stands closer to densities able to survive drought, wildfire and other disturbances over the long term (Young *et al.* 2017). In addition, although fire reduced the amount of surface fuel in the short term, mortality of shrubs (especially on the V1 and V0.5 subplots where the shrub biomass, both dead and live, was at least 50 Mg ha^{-1}; Figs 4, 6, 8)) will add substantially to woody surface fuel loads in the near future as this standing dead material falls to the ground. Managing for low surface fuel loads is especially important when trees are still small (Weatherspoon and Skinner 1995; Lyons-Tinsley and Peterson 2012). Repeated prescribed fires, manual removal or pile burning of dead trees and shrub biomass might be necessary to reduce hazardous fuels and prevent loss of the stand in future wildfires.

After data for the present study were collected, four of the 15 plots (all the plots on the east side of the road in Fig. 1) burned in the 2018 Mendocino Complex Fire, the largest in modern Californian history. Of these four, only the one plot that did not burn during the 2012 Mill Fire burned severely and was a total loss. The plot immediately adjacent on the same slope and aspect – with the same density treatment, which did previously burn in 2012 – remains mostly green with very light tree mortality. The other two plots experienced a patchy burn, with no discernible mortality across all shrub treatments. Interestingly, the majority of dead manzanita stems from the 2012 backfire were still standing, and many were not consumed. Fire appeared to again spread primarily through the light needle litter. This will likely change in future fires as the manzanita stems decay and fall to the ground. In summary, even though the tree densities and landscape context of the four plots varied somewhat, the stark difference in mortality between previously burned and unburned plots highlights the value of surface fuel treatments for enhancing resilience to subsequent wildfires, particularly in young plantation stands.

Conclusions

Although a backfire was ignited during dry midsummer conditions, we showed here that it did not negatively affect the ponderosa pine plantation. Although mortality was highly variable (ranging from nearly 0 to 100%), pre-fire differences in cover of understorey shrubs, tree density, tree basal area and tree height to live crown did not significantly influence post-fire tree mortality. Variation in mortality was likely more the result of ignition patterns during the backfire operation and fuel manipulation (e.g. bulldozer piling adjacent to some subplots) that immediately preceded the burning. Bole char height explained 65% of the variation in mortality rate. Comparing tree rings of surviving trees before and after the fire showed that growth was not significantly influenced by burning. Increase in growth of living trees through the release from inter-tree and shrub competition might have offset any fire-caused growth reduction. The fact that this plantation survived a managed fire in July suggests that wider date, weather and climate windows might exist for using prescribed fire in young stands than those commonly employed today, despite the relative vulnerability of small trees to fire.

Conflicts of interest

The authors declare that there are no conflicts of interest.

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