CSIRO PUBLISHING

International Journal of Wildland Fire **2019**, *28*, 495–511
https://doi.org/10.1071/WF18141

Implementation constraints limit benefits of restoration treatments in mixed-conifer forests

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Abstract. Forest restoration treatments seek to increase resilience to wildfire and a changing climate while avoiding negative impacts to the ecosystem. The extent and intensity of treatments are often constrained by operational considerations and concerns over uncertainty in the trade-offs of addressing different management goals. The recent (2012–15) extreme drought in California, USA, resulted in widespread tree mortality, particularly in the southern Sierra Nevada, and provided an opportunity to assess the effects of restoration treatments on forest resilience to drought. We assessed changes in mixed-conifer forest structure following thinning and understorey burning at the Kings River Experimental Watersheds in the southern Sierra Nevada, and how treatments, topography and forest structure related to tree mortality in the recent drought. Treatments had negligible effect on basal area, tree density and canopy cover. Following the recent drought, average basal area mortality within the watersheds ranged from 5 to 26% across riparian areas and 12 to 44% across upland areas, with a range of 0 to 95% across all plots. Tree mortality was not significantly influenced by restoration treatments or topography. Our results suggest that the constraints common to many restoration treatments may limit their ability to mitigate the impacts of severe drought.

Additional keywords: bark beetles, drought, forest restoration, fuel treatments, mortality, prescribed burning, preventive management, resilience and resistance of ecosystems, Sierra Nevada, treatment effects, vegetation composition and structure.

Received 23 August 2018, accepted 26 March 2019, published online 16 May 2019

Introduction

A common goal in management of western United States (USA) forests is to increase resilience to fire, climate change and other stressors (Reinhardt *et al.* 2008; USDA-USDI 2014). In forests that once experienced a frequent low-severity fire regime but have undergone many decades of fire suppression, goals for increasing resilience include reducing densities of small, shade-tolerant trees and surface fuels, and increasing structural heterogeneity (North *et al.* 2009). Restoration activities in these forests typically include thinning, prescribed fire, or a combination of the two (Stephens *et al.* 2009; Fulé *et al.* 2012). In recent decades, the impetus for restoration was often based on expected fire behaviour (Stephens *et al.* 2013). Dense stands dominated by small, shade-tolerant species are more prone to crown fire that causes near to complete tree mortality and can spread across the landscape, leading to large deforested areas (Lydersen *et al.* 2016; Lydersen *et al.* 2017). A more recent concern is the effect of altered forest structure on resilience to a changing climate characterised by warmer temperatures and increased drought frequency and severity (Stephens *et al.* 2018; Thorne *et al.* 2018).

Although the need for forest restoration is widely recognised, real-world limitations often constrain the extent and intensity of treatments (Collins *et al.* 2010). For treatments involving mechanical thinning, constraints on the extent of treatments include land designation (e.g. wilderness, protected habitat, stream buffer), operability (slope gradient, distance to existing roads) and funding (Fulé *et al.* 2006; North *et al.* 2015), while constraints on treatment intensity are largely a product of residual forest structural guidelines in forest plans (e.g. leaving large trees and moderate to high canopy cover) (Collins *et al.* 2011). Disposing of the unmerchantable material created from treatments is also a challenge (Springsteen *et al.* 2011). The major constraints on implementing fire treatments include hazardous fuel conditions (elevated risk of escape), limited burn windows (from both air quality and fuel moisture) and lack of available resources (e.g. fire crews, engines) (Quinn-Davidson and Varner 2012). The net effect of these constraints is a large discrepancy between the area needing treatment and the area actually receiving treatment, along with limited success of meeting treatment objectives (North *et al.* 2012; Vaillant and Reinhardt 2017).

Increased forest density resulting from fire suppression has increased competition between trees, which can reduce individual tree vigour (Gleason *et al.* 2017). This reduction in tree vigour is exacerbated by prolonged drought, which can considerably limit the ability of individual trees to mount defensive actions against bark beetle attack (McDowell *et al.* 2011). Management actions that reduce stand density can decrease susceptibility to bark beetles. Thinning to reduce tree density can be an effective means to minimise mortality from bark beetles (Fettig *et al.* 2007; Kolb *et al.* 2016). The effect of burning is less clear, because it may increase susceptibility to beetles by damaging tree tissue and causing stress to the tree (Fettig *et al.* 2007). However, prescribed fire has been shown to reduce tree mortality in response to drought, presumably through lowering tree density compared with unburned stands (van Mantgem *et al.* 2016). In addition, low-severity fire can increase tree resistance to future bark beetle attacks (Hood *et al.* 2015).

California experienced an extreme multiyear drought from 2012 to 2015, with anomalously low precipitation coupled with above-average temperatures (Diaz and Wahl 2015; Robeson 2015). An estimated 129 million trees died in California in association with this extreme drought (USDA Forest Service 2017), with the highest rate of mortality observed in 2015. The cause of this widespread tree mortality is thought to be the combined effects of severe moisture stress from drought, decreased tree vigour from elevated tree densities and a coincident bark beetle outbreak (Byer and Jin 2017; Young *et al.* 2017). Although tree mortality was severe and extensive, particularly in the southern and central Sierra Nevada, there was noticeable variability in the patterns of mortality across gradients of elevation, topography, substrate and tree species composition (Paz-Kagan *et al.* 2017). More information is needed on the specific effects of this mortality event on forest structure and composition.

In the present study, we take advantage of a long-term research site in the southern Sierra Nevada, located in an area that was highly affected by drought-related, beetle-induced forest mortality. The Kings River Experimental Watersheds study was installed to investigate the effect of forest restoration treatments on watershed condition and stream hydrology and ecology. Thinning treatments were completed in the first year of the recent severe drought (2012), before the onset of widespread tree mortality throughout California. Burning treatments were applied 1–4 years after the thinning treatments (2013, 2016), allowing analysis of the effect of ongoing management in the face of drought and beetle stress. Here, we assess (1) the effects of the forest restoration treatments on stand structure; (2) what factors, including treatments, affected observed tree mortality; and (3) tree demographics in relation to the observed mortality.

Methods

Study area

Data were collected at the Kings River Experimental Watersheds (KREW) within Sierra National Forest in the southern Sierra Nevada, California (Fig. 1). KREW is a long-term research area established by the US Forest Service to characterise the variability in stream and watershed characteristics for Sierra Nevada headwaters and to evaluate the effects of forest restoration activities (tree thinning and understorey prescribed fire). The greater study consists of two sites, with different elevation ranges, each with four watersheds of first- to third-order streams (Fig. 1, Table 1). The Providence site (D102, P301, P303 and P304) is at the rain–snow transition zone and ranges in elevation from 1485 to 2115 m. The Bull site (B201, B203, B204 and T003) ranges in elevation from 2050 to 2490 m, and precipitation mostly falls as snow. The eight watersheds range in size from 0.5 km^2 (P304 and B201) to 2.3 km^2 (T003). The study area is characterised by a Mediterranean climate with warm, dry summers and cool, wet winters, with nearly all of the precipitation falling between October and May. Annual precipitation is similar at the two sites (Table 2), but the amount of precipitation that falls as snow differs. At the Providence Creek site, 35–60% of precipitation fell as snow between 2004 and 2007, compared with 75–90% at the Bull Creek site (Hunsaker *et al.* 2012). A severe drought occurred in California from water years 2012 to 2015. In addition to lower average annual precipitation during this time period, average minimum and maximum temperature tended to be somewhat higher (Hunsaker and Safeeq 2018) (Table 2).

Vegetation at the Providence Creek site consists of mixed-conifer forest, dominated by white fir (*Abies concolor* (Gordon & Glend.) Hildebr.), incense-cedar (*Calocedrus decurrens* (Torr.) Florin), sugar pine (*Pinus lambertiana* Douglas) and ponderosa pine (*P. ponderosa* Lawson & C. Lawson) (Table 1). The most common hardwood tree species is California black oak (*Quercus kelloggii* Newb.). Vegetation at the Bull Creek site is predominantly red fir (*Abies magnifica* A. Murray bis) forest, consisting mainly of red fir and white fir, with mixed-conifer forest at its lower elevations. Jeffrey pine (*P. jeffreyi* Grev. & Balf.) and lodgepole pine (*P. contorta* Douglas ex Loudon) are also present. The lower-elevation Providence Creek site generally has greater plant species diversity and higher rates of evapotranspiration compared with the Bull Creek site (Safeeq and Hunsaker 2016; Dolanc and Hunsaker 2017). Soils at both sites are coarse and derived from granite (Johnson *et al.* 2011). The historical fire return interval based on data collected at the adjacent Teakettle Experimental Forest, which has an elevation range of 1800–2600 m, was 11–17 years, with the last fire occurring in 1865 (North *et al.* 2005). The Teakettle Experimental Forest is more similar in elevation and forest composition to the upper elevation Bull Creek site used in the present study, which is partially within the Teakettle Experimental Forest. As fire return intervals were historically more frequent (median of 9 years) in dry mixed-conifer than in red fir-dominated forests (Van de Water and Safford 2011), the fire return interval may have been shorter at the lower-elevation Providence Creek site. In the modern fire record (1911–2018) less than 4 ha burned within the KREW.

Treatments

The restoration treatments evaluated were thinning alone, understorey burning alone, thinning combined with understorey burning, and control (no treatment). Once the experimental sites

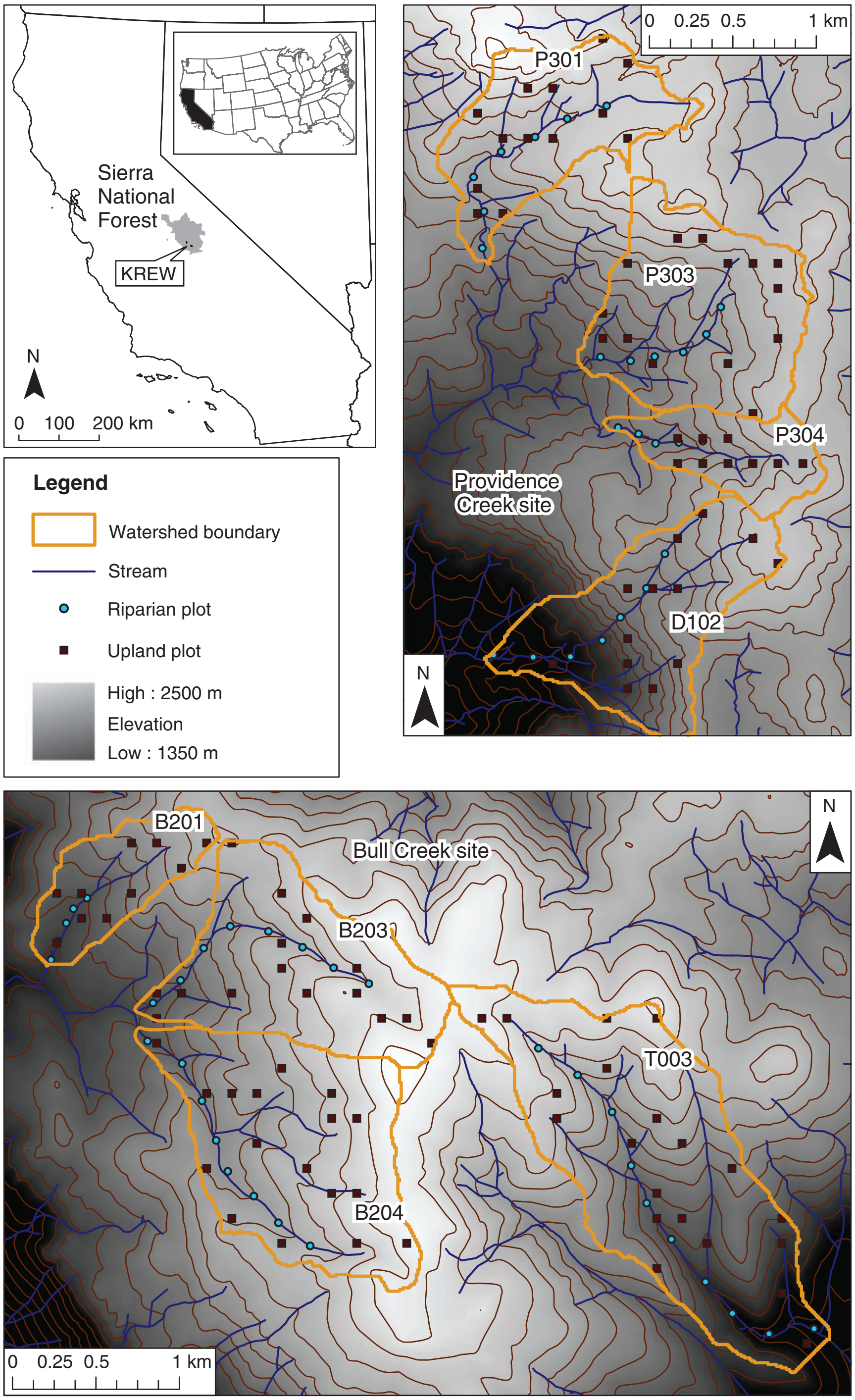

Fig. 1. Map of study area showing the eight watersheds and their location in California, USA. The inset in the upper left panel shows the location of California within the USA (black shading).

were selected, forest managers worked with research staff to determine which watersheds would receive which treatment based on forest condition and wildlife species of concern. For example, P303 had nesting trees for California spotted owl, so it was selected to be burned without thinning; T003 was designated as a control because it is in the Teakettle Experimental Forest, and this watershed was never commercially harvested. Each site, Bull Creek and Providence Creek, contained four watersheds to which one of the four treatments was applied. There were no replicate treatments at the watershed level within a site.

For watersheds designated for thinning, the treatments were applied in the summer of 2012. Thinning was delayed from the

Table 1. Comparison of the eight Kings River experimental watersheds

Mean values among plots are shown, with ranges in parentheses. Aspect was transformed using a cosine function, with 45° set to the maximum value of 2 and 225° set to the minimum value of 0. Tree basal area, density and species composition by basal area represent the pretreatment conditions for trees ≥1 cm diameter at breast height. Species codes are abco, white fir; abma, red fir; cade, incense cedar; pico, lodgepole pine; pila, sugar pine; yp, Jeffrey pine and ponderosa pine; quke, black oak

Watershed	Treatment	*n* riparian	*n* upland	Elevation (m)	Transformed aspect	Live tree basal area (m^2 ha^{-1})	Live tree density (ha^{-1})	Species composition (% basal area)
Providence Creek								
D102	Thin	7	14	1749 (1494–1976)	0.2 (0–1.3)	50 (4.5–119.6)	812 (88–1750)	33% cade, 30% abco, 22% pila, 13% yp, 2% quke
P303	Burn	6	15	1877 (1751–1973)	0.3 (0–1.5)	43.8 (0–97)	613 (0–2038)	51% abco, 42% cade, 7% pila
P301	Thin + Burn	7	13	1952 (1810–2050)	0.6 (0–1.6)	63.4 (3–142.1)	614 (125–1500)	46% abco, 29% cade, 23% pila, 2% yp
P304	Control	5	9	1858 (1777–1947)	0.7 (0–1.9)	44.8 (15.1–147.2)	598 (150–2075)	49% abco, 33% cade, 18% pila
Bull Creek								
B201	Thin	5	10	2230 (2160–2365)	0.2 (0–0.7)	54.1 (0.5–123.5)	648 (100–2225)	57% abma, 22% abco, 11% pila, 10% pico
B203	Burn	8	15	2346 (2208–2457)	0.8 (0–2)	46.2 (0–116.3)	548 (0–2463)	78% abma, 15% abco, 3% pico, 3% yp, 1% pila
B204	Thin + Burn	8	17	2338 (2214–2458)	0.8 (0–2)	54.8 (0–158.2)	592 (0–2425)	93% abma, 6% abco, 1% pila
T003	Control	10	20	2264 (2058–2450)	1 (0–2)	79.2 (0–211.7)	522 (0–2050)	52% abco, 42% abma, 3% pila, 2% cade

planned start in 2007 or 2008 to 2012 by the National Environmental Policy Act process. The thinning treatments included both conventional timber harvest (chainsaw-felling, tops and limbs removed and left in the stand, logs skidded to a landing) in mature stands and precommercial thinning within younger (<30 years old), even-aged stands. Additionally, younger stands with high shrub cover (>50% cover) were masticated to <10% shrub cover. The thinning prescription in mature stands was based on uneven-aged management, which involved tree removal across diameter classes to a target basal area of 27–55 m^2 ha^{-1}. Target basal areas varied based on predetermined aspect and topographic position classes, consistent with the conceptual framework described by North *et al.* (2009). California black oak, sugar pine and ponderosa pine were preferred for retention. Additionally, trees removed from National Forest land had a maximum diameter at breast height (DBH) of 76 cm (30 in), but some trees up to a DBH of 117 cm (46 in) were cut on privately owned Southern California Edison land in the Providence site. During implementation, ~10–25% of the area planned for thinning (or mastication) within treatment watersheds was excluded from operation owing to slope steepness (generally >30–35% slope) and lack of existing roads; this was especially true for D102, the lower-elevation thin-only watershed. Note, although the plan allowed for treatment in these areas, the upper diameter cap and basal area targets limited removals such that the additional measures required for access were deemed economically infeasible. The KREW study was designed to evaluate possible effects of restoration activities that were allowed near the streams; trees could be felled within 15 m (50 ft) of the stream and dragged out, but mechanical equipment was prohibited within 30 m (100 ft) of the stream bank.

Prescribed burns were applied in fall of 2013 at the higher-elevation Bull Creek site, and in fall (autumn) of 2016 at the mid-elevation Providence Creek site. In the Bull site, burning occurred the year after thinning as planned, but in the Providence site, the drought conditions, coupled with the occurrence of a large, long-duration wildfire in 2015 (Rough Fire), delayed the burn for 3 years. The primary objective of the burns was to consume surface and ground fuels, and not necessarily to modify forest structure. Backing fire was the predominant mode of spread, but some flanking and even small runs of head fire did occur (K. Ballard, Sierra National Forest, pers. comm.). Wind speeds were generally <4.5 m s^{-1}, although the plan allowed for speeds up to 8.9 m s^{-1}. Maximum flame lengths were 1.2–1.8 m. Temperature, relative humidity and fuel moisture conditions were generally cool and mild (Table 3). The prescriptions allowed fire to be applied up to 1.5 m (5 ft) of the stream channel but not within green riparian vegetation. 'Best Management Practices' were used according to Forest Service standards (appendix C in USDA 2011). Soil disturbance data from mechanical thinning were collected in 2013, the year following thinning, using a 4-m^2 quadrat placed on a uniform grid (with a random start) with spacing at 150-m across each KREW watershed (Page-Dumroese *et al.* 2009). Each point was visually assessed for the degree of soil disturbance indicators such as forest floor impacts, topsoil displacement, rutting, compaction and soil structure changes, and classified as low (Class 1), moderate (Class 2), high (Class 3), or no evident soil disturbance (Class 0). Soil burn severity was assessed in 2014 at the Bull

Table 2. Climate at the two study sites

Values shown are mean ± s.d. for each time period (Hunsaker and Safeeq 2018). Annual data are shown for water years (1 October through 30 September)

Climate variable	Providence Creek		Bull Creek	
	2005–11	2012–15	2005–11	2012–15
Annual precipitation (cm)	151[A] ± 51	76 ± 19	159[A] ± 51	87 ± 19
Daily minimum temperature (°C)	4.3 ± 0.5	5.8 ± 0.6	1.9 ± 0.5	3.3 ± 0.6
Daily maximum temperature (°C)	14.2 ± 0.7	15.5 ± 0.6	12.1 ± 0.7	13.0 ± 0.4

[A]4 years during this period had above-average precipitation; the 30-year normal (1981–2010) based on PRISM data is 111 cm (Safeeq and Hunsaker 2016).

Table 3. Weather and fuels conditions during the prescribed burns

For weather variables and fuel moisture, averages (and ranges) over all days on which burning was conducted are shown. FM stands for fuel moisture. The fuel loads shown are for pretreatment conditions in 2006. Fuels were measured on two 20-m planar intersect transects (Brown 1974)

	Bull	Providence
Temperature (°C)	7.8 (4.4–11.1)	9.6 (9.4–10)
Relative humidity (%)	36 (10–77)	37 (22–69)
1-h FM (%)	6.2 (2–16)	7.3 (4–14)
10-h FM (%)	8 (4–17)	8.8 (6–14)
100-h FM (%)	15 (10–20)	17.3 (16–18)
1000-h FM (%)	17.9 (17–20)	21.3 (21–22)
Riparian (Mg ha^{-1})		
1-h	0.5 (0–6.2)	0.6 (0–3.7)
10-h	2.1 (0–25.9)	1.9 (0–12.8)
100-h	2.7 (0–27.3)	3.7 (0–17.9)
1000-h	79.5 (0–831.1)	65.4 (0–431.9)
Litter	17 (0–208.7)	36.6 (0–185)
Duff	48.9 (0–417.3)	63.6 (0–236.9)
Upland (Mg ha^{-1})		
1-h	0.8 (0–5.7)	0.8 (0–6.5)
10-h	2.7 (0–15.8)	2.9 (0–15.8)
100-h	3.7 (0–53.6)	3.2 (0–42)
1000-h	52.4 (0–913.4)	47.2 (0–1100.9)
Litter	19.8 (0–222.8)	23 (0–474.1)
Duff	68.2 (0–666.9)	67.1 (0–518.2)

Creek site and in 2017 at the Providence Creek site, 1 year after prescribed burns were conducted. Soil burn severity was measured on the same 150-m grid and 4-m^2 quadrats using the methods of Parson *et al.* (2010). Each point was classified as unburned, low, moderate, or high soil burn severity based on the degree of consumption of soil surface organic layers, condition of roots, the appearance of the exposed mineral soil and the soil structure.

Vegetation sampling

Vegetation plots were established in 2003 before conducting restoration treatments at KREW. Plots were placed in riparian and upland areas within each watershed. The number of plots per watershed (Table 1) was dependent on stream length for riparian and size of the watershed for upland (Dolanc and Hunsaker 2017). The upland plots were located on the same 150-m grid used to sample soil disturbance and burn severity; a random sample of these grid points was chosen in proportion to the size of each watershed. The riparian plots were located every 50 m along alternating sides of each stream. Plots were 10 × 20 m, oriented in a random direction in the upland and oriented perpendicular to the stream in the riparian zone. In each plot, the species and DBH of all live trees ≥1 cm DBH were recorded. Prior to 2017, snags that were intact at DBH were measured but not identified to species. In 2017, the species and decay class of snags were also noted. Cover of all woody species was measured at 1-m intervals along a 20-m transect spanning the centre of the plot using the point-intercept method. Cover, both below and above 2 m in height, was recorded. A densitometer was used to measure cover of woody plants that were >2 m in height. Understorey herbaceous plants and fuels were also measured, but were not assessed in this study.

Vegetation data were collected annually for each plot from 2003 to 2006 before restoration treatments (Table 4). Data from these years were averaged for each plot to estimate the pretreatment condition. A post-thinning vegetation measurement was done on upland plots in both sites in 2013. In 2014, all plots in the Bull Creek site were remeasured (the summer following prescribed burning). The cover of woody species for all plots in the Providence Creek site was also remeasured in 2014, along with the trees in the riparian plots. In 2017, trees and cover of woody species were remeasured for all plots (the summer following prescribed burning for the Providence Creek site, 4 years post burning for the Bull Creek site; Table 4).

Statistical analysis

We assessed changes to forest structure over time with a mixed-model ANOVA using Proc Mixed in *SAS version 9.4*. The response variables assessed were total live basal area, live tree density for trees <15.2, 15.2–30.4, 30.5–50.8 and >50.8 cm DBH, cover of woody plants >2 m in height and cover of woody plants ≤2 m in height. Plots were not treated as independent replicates; watershed was included as a random effect to account for the spatial structure of the data. Year, site, treatment and type of plot (riparian or upland), along with all interactions, were included in the model as fixed effects. All variables were transformed with a square-root function to improve normality of residuals, except trees >50.8 cm and canopy cover >2 m, which were not transformed. Significance was assessed using a significance value adjusted by the Bonferroni correction ($\alpha = 0.0083$ for the upland plots at the Bull Creek site that had four time points of data and $\alpha = 0.0167$ for all other plots that had three time points of data).

Table 4. Timeline of sampling and treatments

x indicates that data was collected that year. Vegetation sampling was not continuous owing to delays in treatments

Site	Type	Measurement	2003	2004	2005	2006	2012	2013 Summer	2013 Fall	2014	2016 Fall	2017
Bull	Riparian	Trees	x	x	x	x	Thin		Burn	x		x
Bull	Riparian	Cover	x	x	x	x	Thin		Burn	x		x
Bull	Upland	Trees	x	x	x	x	Thin	x	Burn	x		x
Bull	Upland	Cover	x	x	x	x	Thin	x	Burn	x		x
Providence	Riparian	Trees	x	x	x	x	Thin			x	Burn	x
Providence	Riparian	Cover	x	x	x	x	Thin			x	Burn	x
Providence	Upland	Trees	x	x	x	x	Thin	x			Burn	x
Providence	Upland	Cover	x	x	x	x	Thin	x		x	Burn	x

We assessed factors related to the proportion and number of dead trees in 2017 with either a generalised linear mixed model or a mixed model, using Proc Glimmix and Proc mixed respectively. For the proportion of dead BA, Proc Glimmix was used to account for the β distribution of the response variable. Proc mixed was used for total basal area of dead trees and total density of dead trees. As in the model to assess changes in forest structure over time, plots were not treated as independent replicates; both types of model included watershed as a random effect to account for the spatial structure of the data. Fixed effects included site, treatment, type, type × treatment, elevation, aspect, slope steepness and the live tree density following the thinning treatments (2013 for upland plots, 2014 for riparian plots). Aspect was cosine transformed to range from 0 to 2, with lower values corresponding to more south-westerly aspects ($0 \approx 225°$; $2 \approx 45°$). Basal area and density of dead trees were square-root-transformed to better meet model assumptions. Because we were primarily interested in mortality of large trees, 28 plots that had no trees >50.8 cm DBH were excluded from the analysis of the factors contributing to mortality.

Results

Treatment effects

The restoration treatments at KREW were very light. Gridded sampling across the treated watersheds showed that 48% of the area within thinned watersheds at the Bull Creek site had no soil disturbance after thinning, while 23% had high, 13% had moderate, and 17% had low soil disturbance. Soil disturbance was even lower at the Providence Creek site, with 71% of the area within thinned watersheds showing no soil disturbance after thinning, while 9% had high, 4% had moderate, and 17% had low soil disturbance. After prescribed fire, 61% of the Bull Creek watersheds where fire was applied appeared unburned. Soil burn severity in these watersheds was low for 34% and moderate for 5% of the area. Fire effects were slightly greater in the burned watersheds at the Providence Creek site, although 48% of the area appeared unburned. Soil burn severity in the Providence Creek watersheds was low for 37%, moderate for 14% and high for 1% of the area. Thus, treatments affected approximately 50% or less of the area within the watersheds where they were applied.

Based on the record of trees included in the timber sale (those ≥25.4 cm DBH), there were ~5116 trees of merchantable size removed from the two thinned watersheds at the Providence Creek site, and 7161 trees removed from the two thinned watersheds at the Bull Creek site (K. Ballard, Sierra National Forest and R. Stewart, Southern California Edison, pers. comm.). Averaged over the thinned watershed area, this gives a watershed-level treatment intensity of 23 trees ha^{-1} and 4.5 m^2 ha^{-1} basal area removed from the Providence Creek watersheds and 33 trees ha^{-1} and 4.6 m^2 ha^{-1} basal area removed from the Bull Creek watersheds. Note that small trees (<25.4 cm DBH) are not included in these estimates and that the intensity was higher in some areas because the area used to calculate this overall average includes steep slopes and plantations that were not commercially thinned. Despite these overall reductions, there were no significant effects on basal area from the thinning treatments as measured by the vegetation plots (Fig. 2). The thinning treatments also had a negligible effect on tree density (Fig. 3). The only significant difference detected in tree density between the pretreatment and post-thinning time points was a decrease in the density of trees <15.2 cm DBH in the riparian plots in P303, the mid-elevation burn-only watershed. As no management activities occurred on P303 before the post-thinning data collection in 2014, this is likely due to slight variation in the trees measured at each grid location in different years, or possibly due to tree mortality or trees growing into a different size category.

Burning also had little effect on tree density or basal area, although the effects of the prescribed fires on mortality are more difficult to disentangle from the effects of the drought because the fires were implemented after the onset of the drought. For Bull Creek, the upper-elevation watersheds, the only significant effect that may be due to prescribed fire was significantly fewer trees <15.2 cm DBH in the upland plots of the thin + burn watershed, B204, in 2014 following implementation of the burn (Fig. 3). There was a slightly greater number of significant changes in the mid-elevation watersheds of Providence Creek that were burned in 2016, which was later in the drought period. Total basal area in upland plots in the thin + burn watershed, P301, was significantly lower in 2017 than that measured post thinning in 2013, but not significantly different than the pretreatment value (Fig. 2). The plots in P301 also showed some reductions in tree density, with significantly fewer riparian trees <15.2 cm DBH and fewer upland trees 30.5–50.8-cm DBH in 2017 as compared with the pretreatment forest, and fewer upland trees <15.2 cm DBH in 2017 compared with both prior

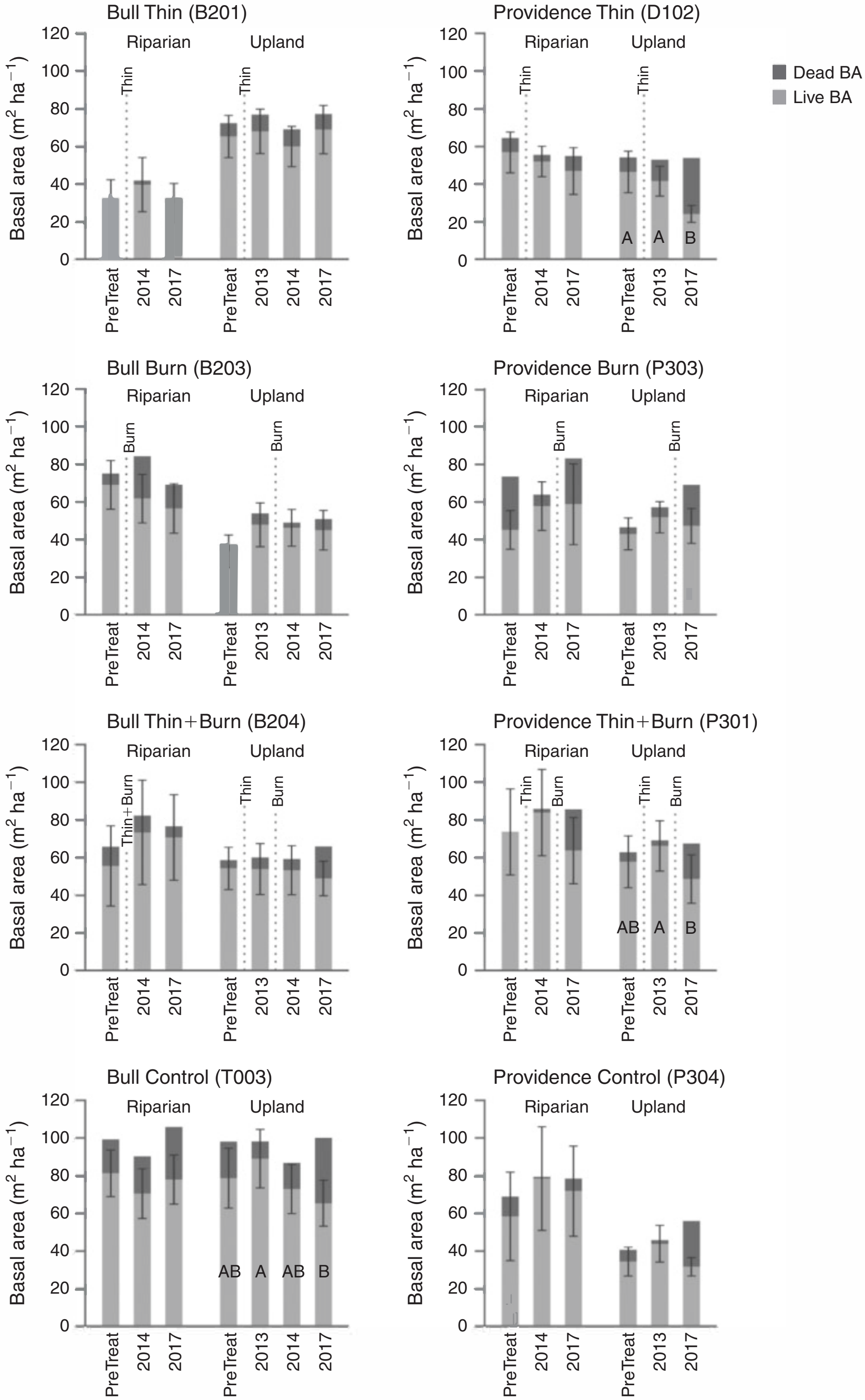

Fig. 2. Change in mean tree basal area (BA) over time for trees ≥1 cm diameter at breast height (DBH). The pretreatment time point is the average BA of conifer and oak trees recorded for each plot in 2003–06. Thinning treatments were applied in summer of 2012. The burning treatments were applied in fall of 2013 in the high-elevation watersheds and in fall of 2016 for the mid-elevation watersheds. Differing letters represent significant differences in live BA between years within a treatment and plot type. Error bars show the standard error for live BA.

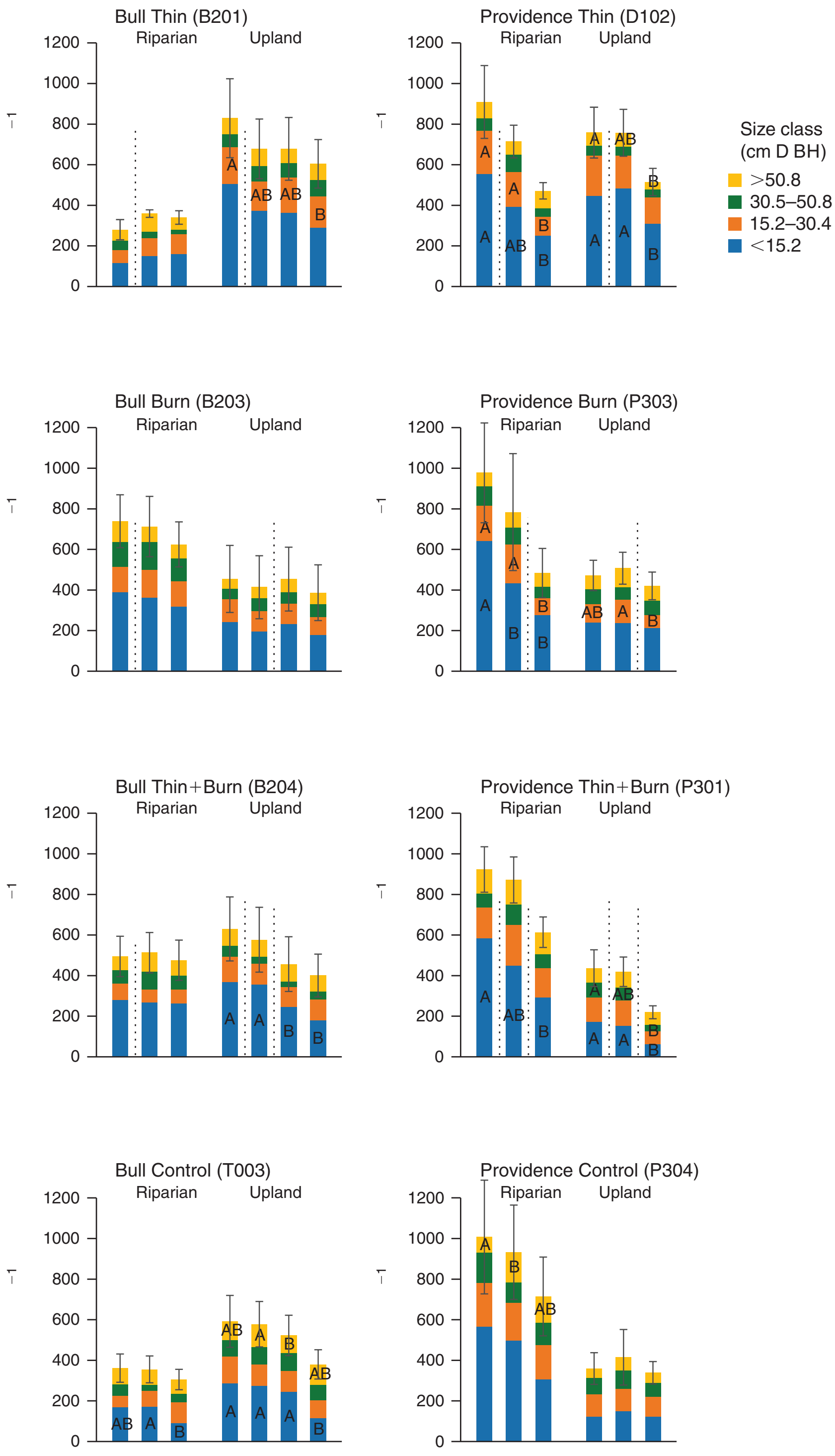

Fig. 3. Change in mean tree density for trees ≥1 cm diameter at breast height (DBH) by size class over time. Size classes are in centimetres DBH. The pretreatment time point is the average tree density of conifers and oak trees recorded for each plot in 2003–06. Thinning treatments were applied in summer of 2012. The burning treatments were applied in fall of 2013 in the high-elevation watersheds and in fall of 2016 for the mid-elevation watersheds. Differing letters represent significant differences between years within a treatment and plot type. Error bars show the standard error for total live tree density.

measurements. The burn-only watershed, P303, had significantly lower density of trees 15.2–30.4-cm DBH in the upland plots in 2017 relative to 2013, but not relative to pretreatment. For the riparian plots in this watershed, the number of trees 15.2–30.4-cm DBH was significantly lower in 2017 relative to both the pretreatment measurement and the post-thinning measurement of 2014 (Fig. 3).

Not surprisingly, treatments had no effect on canopy cover (Fig. 4). In fact, the only significant changes in canopy cover >2 m in height over time were increases. There were a greater number of significant differences between time points for the woody cover ≤2 m in height (Fig. 5). Thinning reduced cover ≤2 m in the upland plots for three of the four watersheds that were thinned, with significant differences between pretreatment and post-thinning time points for B201, B204 and D102. Thinning also reduced cover ≤2 m in the riparian plots for both thinned mid-elevation watersheds (D102 and P301). Cover ≤2 m in 2014 was also significantly reduced compared with pretreatment in both types of plots for the burn-only mid-elevation watershed (P303) in the absence of any management actions. Burning did not have a notable effect on woody cover ≤2 m; there were no significant reductions following the prescribed burns, and cover ≤2 m actually increased in the mid-elevation burn-only upland plots (P303; Fig. 5).

Mortality

Across all plots, the percentage of tree basal area that was dead in 2017 ranged from 0 to 95% (Table 5). Mortality was highly variable across both study sites, and none of the variables considered in the models had a significant effect on mortality, with the exception of a significant positive relationship between density of live trees after thinning and density of dead trees in 2017 (see Table S1 available as Supplementary Material to this paper). As we did not find a significant effect of treatment on mortality, we conclude that the observed mortality can be attributed to the drought. Note that we did not determine the causal factors for tree mortality. Although not significant, in general there was a greater percentage mortality in the upland compared with the riparian plots, and in the mid-elevation Providence Creek watersheds compared with the high-elevation Bull Creek watersheds. The watershed with the greatest proportional basal area mortality was D102 (44%), the mid-elevation thin-only watershed. This watershed had the lowest average and minimum elevation, was one of the most south-westerly facing watersheds, and had the greatest tree density even after implementation of the thinning treatments (Fig. 1, Fig. 3, Table 1). However, neither elevation nor aspect came out as significant in the tree mortality models, and only post-thin live tree density was significantly associated with dead tree density in 2017. Among the higher-elevation watersheds, the control watershed (T003) had the greatest percentage mortality. This watershed was also relatively low in elevation compared with the other Bull Creek watersheds (Table 1), and had many south-facing plots (Fig. 1).

Trees that were dead in 2017 varied by species and size class (Fig. 6). At the Bull Creek site, greater mortality in the upland plots occurred for white fir, which had a greater relative dominance in the control watershed (T003) compared with the other upper-elevation watersheds. Dead trees in the riparian plots at the Bull Creek site were mainly red fir. For both red and white fir, most of the basal area in these species at the Bull Creek site was still alive in 2017. Although less common than fir species, sugar pine, Jeffrey pine and lodgepole pine present at the Bull Creek site were mostly still alive in 2017. Small dead trees also tended to be more abundant than large dead trees at Bull Creek in 2017 (Fig. 6). Despite ~5–30% of trees being dead in 2017 (Fig. 6, Table 5), there were few significant differences to forest structure in 2017 compared with previous years (Fig. 2, Fig. 3), with the most notable difference being significantly fewer trees <15.2 cm DBH in both riparian and upland plots in T003, the control watershed.

At the mid-elevation site (Providence Creek), sugar pine mortality was particularly high (Fig. 6). With the exception of the thin + burn watershed (P301), which had the highest mean and maximum elevation at the Providence Creek site, nearly all the sugar pine died in the upland plots, and approximately half died in the riparian plots. Ponderosa pine was also strongly affected by the mortality episode, with over half of the basal area of that species dead in 2017. In contrast, incense-cedar had very low mortality. White fir had a fair amount of mortality, but most were still alive in 2017. Similar to the Bull Creek watersheds, significant differences in forest structure between 2017 and previous years tended to be modest (Fig. 2, Fig. 3). The thin-only watershed (D102) had significantly lower basal area in 2017 (Fig. 2). This watershed also had fewer trees <15.2 cm DBH in the upland plots and 15.2–30.4 cm DBH in the riparian plots in 2017 compared with all previous years, and fewer trees <15.2 cm DBH in the riparian plots relative to the pretreatment measurement (Fig. 3). This same pattern of fewer small trees was also present in the thin + burn and burn-only watersheds (P301 and P303). For larger trees, two watersheds showed gradual decreases in density over time so that the value in 2017 was significantly different than pretreatment but sequential measurements were not significantly different. This was observed for trees >50.8 cm DBH in upland plots of the thin watershed (D102) and trees 30.5–50.8 cm DBH in upland plots of the thin + burn watershed (P301; Fig. 3).

Discussion

Our study found little effect of treatments on forest structure at KREW. The tree thinning operations ultimately carried out at KREW were constrained in both the areal extent and the intensity (i.e. removals), relative to what was initially planned (R. Rojas, Sierra National Forest, pers. comm.). The discrepancies between planned and implemented treatments were largely due to concerns over potential impacts to individual species habitat, namely California spotted owl (*Strix occidentalis occidentalis*), Pacific fisher (*Pekania pennanti*) and Yosemite toad (*Anaxyrus canorus*). Goals of increasing forest resilience and conserving wildlife habitat are often seemingly in direct conflict, as forest restoration involves removing trees in the mid- to lower canopy strata and creating openings in the canopy. Such changes could adversely affect 'old forest' species that prefer multilayered, closed-canopy conditions (Tempel *et al.* 2015). Concerns over the population sizes and trajectories of these species have forced a fairly conservative approach towards forest restoration on US Forest Service lands, both in terms of overall treatment extent and treatment intensity (Stephens *et al.* 2016). Interestingly, the lack of empirical

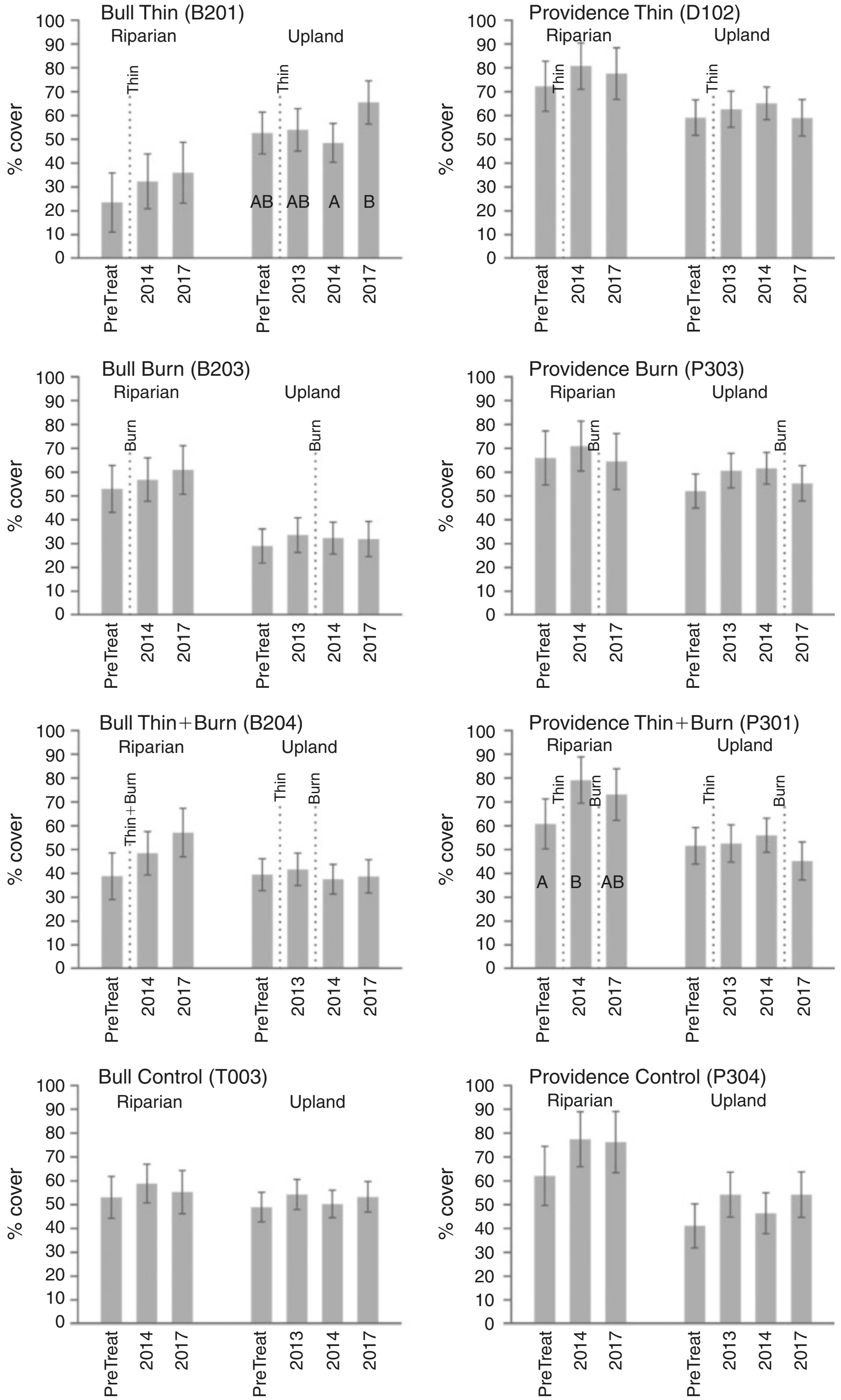

Fig. 4. Change in mean cover of woody plants >2 m in height over time. The timing of the thinning and burning treatments is shown by the dotted lines. Error bars show the standard error. Differing letters represent significant differences between years within a treatment and plot type.

information on the impacts of forest restoration to habitat and other ecosystem values is one factor that limits treatment implementation (i.e. more research is needed).

There can be strong opposition to forest restoration treatments, even for research studies that have a small footprint on the landscape. The treatments at KREW were a glaring example

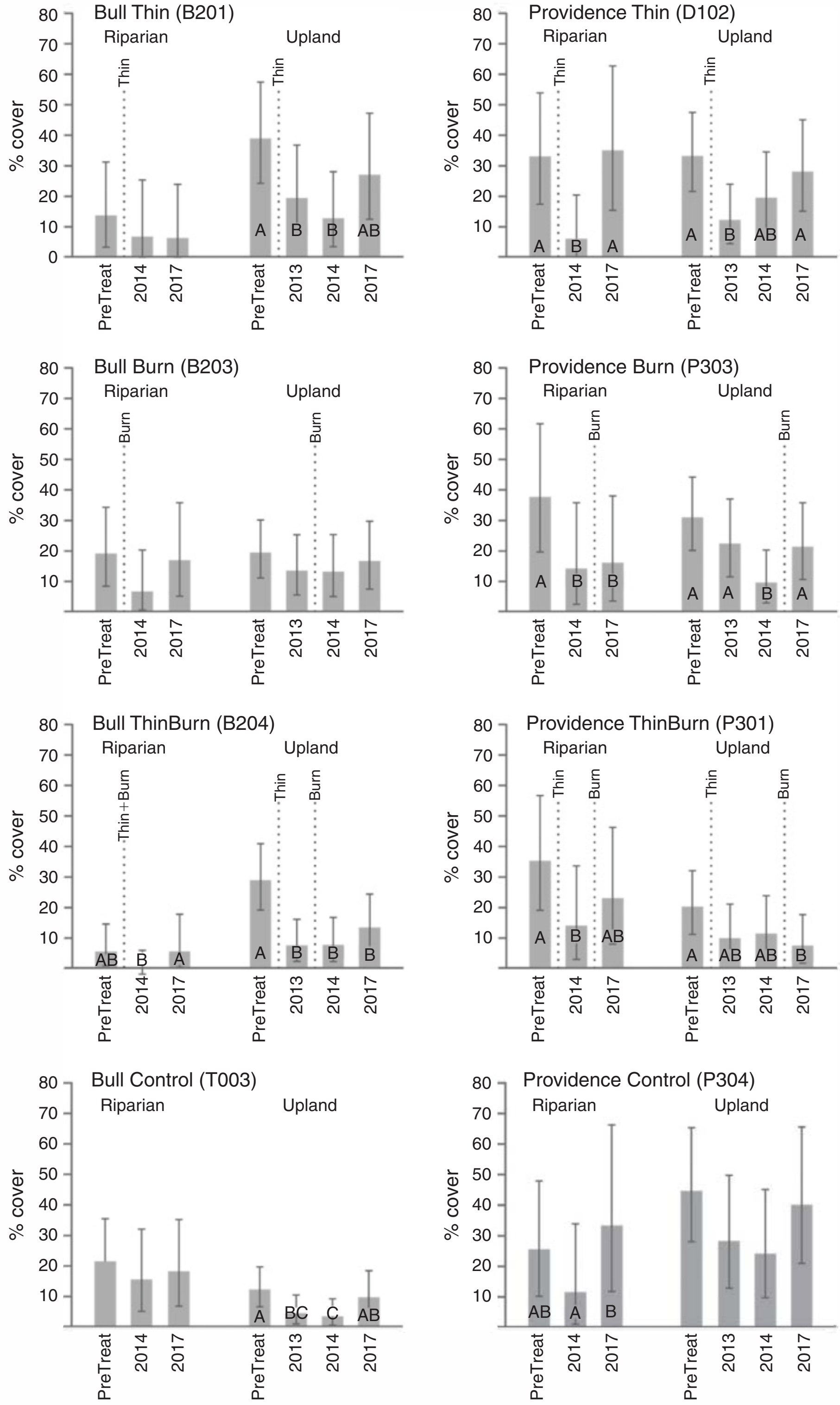

Fig. 5. Change in mean cover of woody plants ≤2 m in height over time. The timing of the thinning and burning treatments is shown by the dotted lines. Error bars show the standard error. Differing letters represent significant differences between years within a treatment and plot type.

Table 5. Percentage of all trees that were dead in 2017

Watershed	Treatment	Type	Dead stems (%) Average	Dead stems (%) Range	Dead basal area (%) Average	Dead basal area (%) Range
Providence Creek						
D102	Thin	Riparian	32.0	(14–61)	21.4	(3–42)
		Upland	29.6	(0–80)	43.5	(0–95)
P303	Burn	Riparian	24.5	(0–51)	24.4	(0–79)
		Upland	21.6	(0–60)	25.9	(0–90)
P301	ThinBurn	Riparian	24.0	(0–50)	14.8	(0–72)
		Upland	35.1	(0–73)	24.7	(0–87)
P304	Control	Riparian	10.2	(0–28)	12.3	(0–59)
		Upland	19.2	(0–42)	27.3	(0–80)
Bull Creek						
B201	Thin	Riparian	7.8	(0–22)	5.5	(0–16)
		Upland	16.2	(0–45)	11.9	(0–73)
B203	Burn	Riparian	6.9	(0–33)	14.9	(0–80)
		Upland	19.8	(0–67)	11.5	(0–82)
B204	ThinBurn	Riparian	10.4	(0–45)	4.5	(0–28)
		Upland	25.4	(0–63)	15.1	(0–70)
T003	Control	Riparian	33.3	(0–63)	25.9	(0–84)
		Upland	33.6	(0–83)	30.6	(0–83)

of this; the Environmental Impact Statement (USDA 2011) was appealed and litigation was threatened, resulting in a delay to treatment implementation (Table 4). Following this appeal, the maximum diameter limit for tree harvest was restricted on Forest Service land to 50.8 cm in D102 and 76.2 cm in the other thinned watersheds. The original diameter limit proposed for thinning was 88.9 cm. These diameter restrictions were in addition to the existing requirement to retain >50% canopy cover in several units for Pacific fisher rest sites (USDA 2011). Economic limitations resulting from these constraints, along with concerns of potential impacts to soil and aquatic ecosystems limited temporary road construction, removal of trees by helicopter, and other mitigation measures to access more difficult ground (e.g. moderately steep slopes, 30–45%). These limitations further contributed to the discrepancies between planned vs implemented by excluding portions of the treated watersheds from thinning. Beyond the impediments to thinning implementation, the prescribed burns at KREW were predominantly focused on modifying understorey conditions, i.e. underburning. Lower-intensity burns were planned owing to the potential for escape associated with the high surface and ladder fuel loads (Table 3), and concerns over potential exacerbated large-tree mortality brought about by the severe drought. As with the thinning, the end result was an overall lack of significant forest structural changes.

Our study found no effect of either thinning or burning treatments on tree mortality associated with the drought. Bark beetle mortality following fire can be elevated compared with that following thinning (Maloney *et al.* 2008), although beetle activity may be greater in untreated areas relative to treated (Roccaforte *et al.* 2018). In contrast to our results, Knapp *et al.* (2017) found that prescribed burning conducted in 2014 in a mixed-conifer forest in the central Sierra Nevada, which coincided with the same severe regional drought, increased bark beetle-related mortality relative to the thin-only and control treatments, with an average of 13.3 and 2.6% mortality in burned and unburned units respectively. Although their study site had similar forest composition to the lower-elevation watersheds at KREW, overall mortality was much lower at their study site compared with what was observed at KREW (Table 5), in line with the overall trend of the greatest tree mortality attributed to the 2012–15 drought occurring in the southern Sierra Nevada (USDA Forest Service 2017). Thinning was found to increase drought resistance in the recent drought in mixed-conifer forest in the Klamath mountains in northern California in thinned stands that had a 34% reduction in basal area (Vernon *et al.* 2018). This effect is consistent with the understanding that thinning reduces competition and leads to increased individual tree vigour (Collins *et al.* 2014; van Mantgem *et al.* 2018). However, this increased vigour is likely not an immediate effect, as it may take multiple years for trees to respond to the improved growing conditions. This potential lag in individual tree growth responses may be particularly important in explaining our lack of treatment effect on tree mortality because the timing of treatments coincided with the onset of an extreme multiyear drought in the southern Sierra Nevada. This timing, combined with the lack of noticeable forest structural change from any of the three treatment types employed at KREW, likely overrode any subtle changes in the competitive environment resulting from the treatments.

We also found no effect of aspect or slope steepness on mortality, in contrast to other studies assessing patterns of drought mortality in Sierra Nevada mixed-conifer forests (Guarín and Taylor 2005; Van Gunst *et al.* 2016; Paz-Kagan *et al.* 2017). Two of these studies were conducted over a wider elevational range than our study site and included forest of similar composition to that at KREW in addition to lower- and higher-elevation forest types (Van Gunst *et al.* 2016; Paz-Kagan *et al.* 2017), and one was conducted within the mixed-conifer zone with approximately similar composition to the

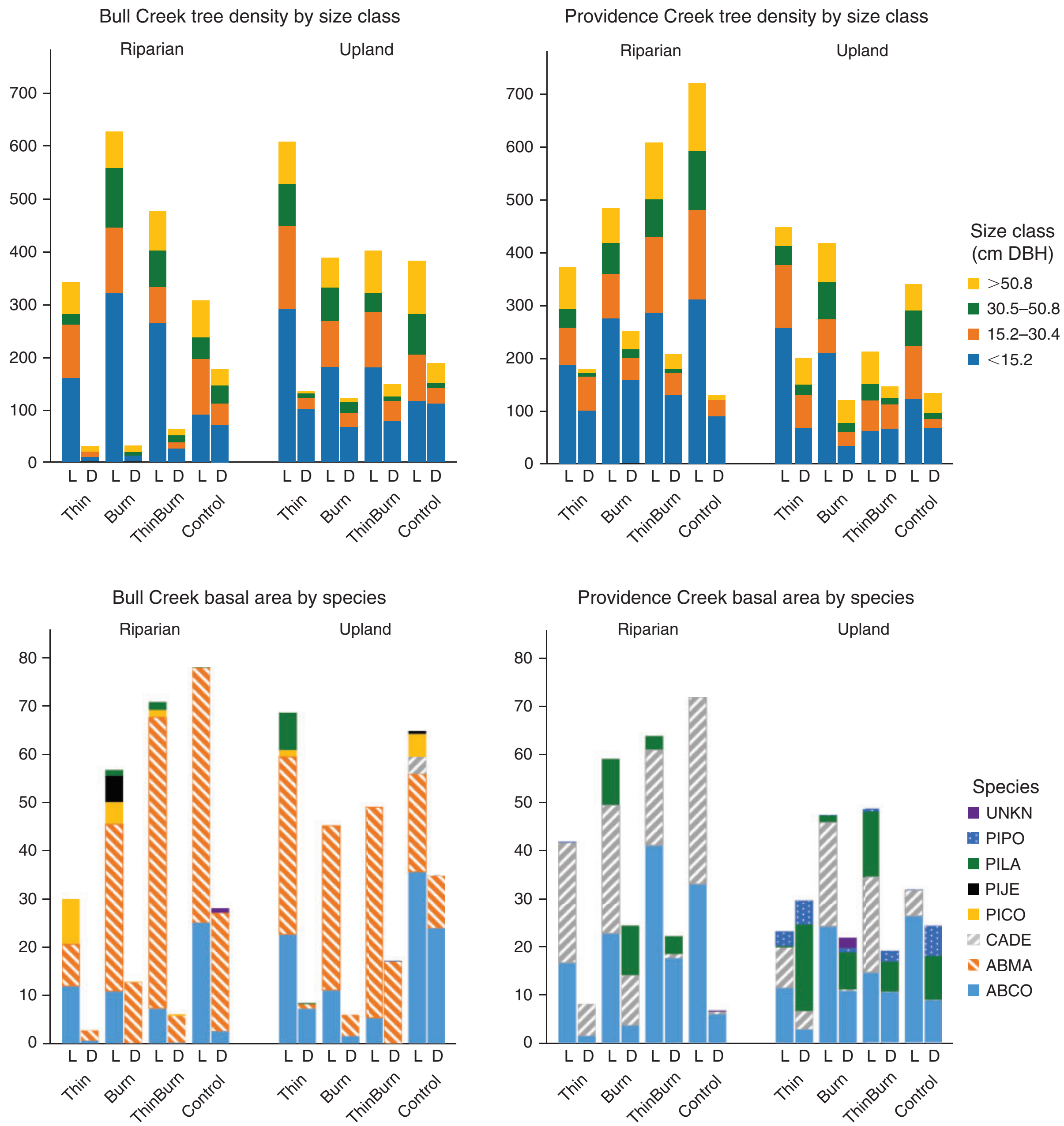

Fig. 6. Live and dead tree density and basal area in 2017. L stands for live trees, D stands for dead trees. Size classes are shown by centimetres diameter at breast height (DBH). Species are white fir (ABCO), red fir (ABMA), incense-cedar (CADE), lodgepole pine (PICO), Jeffrey pine (PIJE), sugar pine (PILA), ponderosa pine (PIPO), and unknown (UNKN).

lower-elevation watersheds at KREW (Guarín and Taylor 2005). Guarín and Taylor (2005) and Van Gunst *et al.* (2016) were also analysing mortality that occurred in droughts before 2010, which were generally less extreme than the 2012–15 drought addressed in the present study. Perhaps our results differ owing to the smaller study site and more extreme drought conditions present in our study. Although not significant, there appeared to be lower mortality in the riparian areas relative to the upland at KREW, particularly for large trees in the mid-elevation Providence Creek site (Fig. 6). Paz-Kagan *et al.* (2017) also found greater mortality at increasing distances from water in nearby Sequoia Kings Canyon National Park.

Although observed mortality was generally greater in the mid-elevation watersheds than the upper-elevation watersheds, elevation was also not a significant predictor of dead tree proportion or abundance at KREW. Other studies of the recent drought effects in the Sierra Nevada found significantly greater tree mortality at lower elevations (Byer and Jin 2017; Paz-Kagan *et al.* 2017). Perhaps this effect is more apparent when elevations lower than those that existed in our study area are included for comparison so that a greater range in elevation is assessed. For example, Paz-Kagan *et al.* (2017) studied mortality along an elevational gradient ranging from 1000 to 3400 m, whereas our study site only ranged from ~1500 to 2500 m. Fettig *et al.* (2019) found significantly higher mortality in the lowest-elevation band (914–1219 m) compared with the highest-elevation band (1524–1849 m) of their study, but neither band was significantly different from their middle-elevation band (1219–1524 m).

Although the watershed with the greatest tree density had the greatest tree mortality (D102), our study found no significant effect of live tree density on either the proportion or amount of dead basal area in 2017. We did find a significant relationship between the number of snags in 2017 and the density of live trees after thinning, but this may just reflect the fact that a large number of dead trees can only occur in areas that had a high tree density to begin with. Fettig *et al.* (2019) similarly found a significant positive relationship between the number of trees killed and tree density in the same 2012–15 drought in ponderosa pine-dominated areas of the Sierra Nevada, but they reported a negative relationship between the proportion of trees killed and both tree density and the stand density index. The lack of relationship between basal area mortality and live tree density in our study was surprising given that other studies have found mortality in the recent drought was higher in areas with greater tree biomass (Guarín and Taylor 2005; Smith *et al.* 2005; Byer and Jin 2017; Young *et al.* 2017). Perhaps the lack of an effect on basal area mortality in our study comes from the generally high tree density found across KREW, and the severity of the 2012–15 drought. Byer and Jin (2017) and Young *et al.* (2017) used regional datasets that included a wider range of forest conditions in the 2012–15 drought than that found at our study site. Our results likely differ from that of studies looking at earlier droughts (Guarín and Taylor 2005; Smith *et al.* 2005) owing to the higher intensity of the 2012–15 drought relative to previous droughts, as severe drought stress may affect a greater number of trees and override the effect of relative competition (Fettig *et al.* 2019).

The patterns of observed mortality across the studied watersheds may have been influenced by species composition, which is influenced by the interaction of factors such as elevation, aspect and soil properties, creating specific growing site conditions (Stephenson 1998). Prior to this recent severe drought, a study in the nearby Teakettle Experimental Forest, which consists of old-growth forest similar in forest composition to the upper elevation Bull Creek site of the present study, found that tree mortality was disproportionately shifted towards large trees, but that there was no noticeable effect on a particular species, i.e. tree species died in proportion to their abundance (Smith *et al.* 2005). In our study, the upper-elevation control watershed (T003) had the greatest mortality among the upper-elevation watersheds. However, most of the dead trees in these watersheds were small white firs, and the control watershed had the highest proportion of white fir. Given the lack of a strong treatment effect on forest structure in these watersheds, it is likely that higher mortality in the control watershed was driven by the greater proportion of white fir, rather than the lack of treatment. This finding was similar to that from a study in ponderosa pine and mixed-conifer forests of the south-west US found that mortality was related to species composition but not stand density, with large white fir and aspen being most susceptible to drought (Ganey and Vojta 2011). In a drought-induced fir engraver outbreak in 1987 in the Lake Tahoe Basin, a greater proportion of fir within a stand was related to greater mortality (Ferrell *et al.* 1994). Although we did not determine the ultimate cause of tree mortality in our study, taken together, these observations emphasise the importance of host-specific agents (i.e. bark beetles) in widespread mortality episodes, rather than simply moisture stress alone brought about by drought.

In contrast to the small-tree and white fir mortality in the upper-elevation watersheds, the mid-elevation Providence Creek site was more likely to have pine and large-tree mortality. This is similar to results from an earlier study in Yosemite National Park that found that most drought-killed white fir were small trees, whereas pines of all sizes were susceptible to drought mortality (Guarín and Taylor 2005). The loss of pines at the lower-elevation site in the recent drought has also been observed elsewhere in the southern Sierra Nevada (Paz-Kagan *et al.* 2017). The greater survival of sugar pine and Jefferey pine in the upper elevation watersheds is likely due to these species being at their upper elevational range, which perhaps allowed some landscape-level discontinuity in pine hosts, and thereby limited the impact of host-specific bark beetles (Seidl *et al.* 2016). Although currently a small component of the forest at the higher elevation, perhaps these species will be able to migrate upslope as they are experiencing greater mortality at low- to mid-elevation ranges in the Sierra Nevada. Lodgepole pine also had high survival, indicating it may have a stable population at our site with respect to drought. At KREW, lodgepole pine mainly occurs at the transition from open meadow habitat to mixed-conifer forest where moisture is readily available.

Treatments did not decrease canopy cover, an important habitat quality for some notable species of concern mentioned previously, the California spotted owl and the Pacific fisher. Two watersheds actually showed slight significant increases in tree cover. These slight increases could be due to tree growth, which may be augmented by removal of understorey trees. As of 2017, there were also no significant changes to canopy cover from the drought. However, our study did not differentiate between live and dead trees in measuring canopy cover. In the watersheds with higher proportions of snags in 2017 such as D102, as needles drop and snags decompose, the overstorey canopy will likely become more open. The effects of the recent mortality episode on wildlife are still unknown at this point. The loss of large trees, which was more common at the mid-elevation site at KREW, has been associated with declining population levels of the California spotted owl (Jones *et al.* 2018). For areas where a greater number of small trees were lost and the large trees providing overstorey cover remain, there may be less of a negative effect because owls prefer large trees and avoid high densities of smaller trees (North *et al.* 2017).

Forests on the west slopes of the Sierra Nevada are vulnerable to climate change, and management will likely need to adapt as climate affects forests (Thorne *et al.* 2018). If the occurrence of severe multiyear droughts increases in the future, bark beetle activity and associated tree mortality can be expected to increase (Guarín and Taylor 2005; Kolb *et al.* 2016). Although the forest restoration treatments at KREW were designed to improve forest resiliency, they were constrained by concerns for terrestrial and aquatic species and implementation conditions. A range of 23 to 33 trees were removed per hectare and 39 to 51% of the watersheds had understorey fire, but the data show little effect of these treatments on forest structure. The lack of substantial forest structural change, and hence limited realised benefit, from similar 'minimal'-impact treatments has been noted in south-western USA forests (Fulé *et al*. 2006; Huffman *et al.* 2018). Forest restoration treatments have the potential to limit the impacts of future droughts, and their interactions with other disturbances, i.e. wildfire (Stephens *et al.* 2018). However, it is clear that under current planning, economic and social constraints, the extent to which restoration treatments are being implemented and the intensity of treatments such as those studied here are likely not enough to actually mitigate these impacts (Stephens *et al.* 2016).

Conflicts of interest

The authors declare no conflicts of interest.

Acknowledgements

Many thanks to Jim Baldwin for his assistance with statistical analysis. We thank Chris Dolanc for establishing the vegetation plots, and E. Adams, A. Armstrong, A. Brandstetter, S. Buchanan, A. Casas, S. Calloway, R. Gallegos, J. Hsu, M. Jabis, R. Koons, J. Musso, J. Olin, J. Palmer, B. Serna, L. Sutherland and C. Unger for collecting the field data. We are grateful to Kevin Mazzocco and Mohammad Safeeq for their help with supervision and training of the 2017 field crew. We thank Ramiro Rojas and Carolyn Ballard for providing information on treatment specifications. KREW was established with USDA Forest Service funding from the National Fire Plan and consistently supported by the Pacific Southwest Research Station, Albany, CA. We received some support from California's State Water Resources Control Board, through Proposition 50 (Water Security Clean Drinking Water, Coastal, and Beach Protection Act of 2002).

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