Forest Ecology and Management 379 (2016) 265–272

Contents lists available at ScienceDirect

Forest Ecology and Management

journal homepage: www.elsevier.com/locate/foreco

Duration of fuels reduction following prescribed fire in coniferous forests of U.S. national parks in California and the Colorado Plateau

Phillip J. van Mantgem [a,*], Laura B. Lalemand [a], MaryBeth Keifer [b], Jeffrey M. Kane [c]

[a] U.S. Geological Survey, Redwood Field Station, Arcata, CA 95521, USA
[b] National Park Service, National Interagency Fire Center, Boise, ID 83705, USA
[c] Department of Forestry and Wildland Resources, Humboldt State University, Arcata, CA 95521, USA

ARTICLE INFO

Article history:
Received 9 May 2016
Received in revised form 18 July 2016
Accepted 20 July 2016
Available online 27 August 2016

Keywords:
Fire effects
Fire modeling
Forest management
Fuels treatments

ABSTRACT

Prescribed fire is a widely used forest management tool, yet the long-term effectiveness of prescribed fire in reducing fuels and fire hazards in many vegetation types is not well documented. We assessed the magnitude and duration of reductions in surface fuels and modeled fire hazards in coniferous forests across nine U.S. national parks in California and the Colorado Plateau. We used observations from a prescribed fire effects monitoring program that feature standard forest and surface fuels inventories conducted pre-fire, immediately following an initial (first-entry) prescribed fire and at varying intervals up to >20 years post-fire. A subset of these plots was subjected to prescribed fire again (second-entry) with continued monitoring. Prescribed fire effects were highly variable among plots, but we found on average first-entry fires resulted in a significant post-fire reduction in surface fuels, with litter and duff fuels not returning to pre-fire levels over the length of our observations. Fine and coarse woody fuels often took a decade or longer to return to pre-fire levels. For second-entry fires we found continued fuels reductions, without strong evidence of fuel loads returning to levels observed immediately prior to second-entry fire. Following both first- and second-entry fire there were increases in estimated canopy base heights, along with reductions in estimated canopy bulk density and modeled flame lengths. We did not find evidence of return to pre-fire conditions during our observation intervals for these measures of fire hazard. Our results show that prescribed fire can be a valuable tool to reduce fire hazards and, depending on forest conditions and the measurement used, reductions in fire hazard can last for decades. Second-entry prescribed fire appeared to reinforce the reduction in fuels and fire hazard from first-entry fires.

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1. Introduction

Following a century of fire exclusion, the reduction of accumulated fuels is a key management goal in fire-prone forest types in the western U.S. (Allen et al., 2002; Brown et al., 2004; Agee and Skinner, 2005; Noss et al., 2006). Prescribed fire, where fire is intentionally ignited under specified conditions (sometimes following thinning treatments), is a common tool used to remove surface fuels and reduce the continuity of live fuels (shrubs, small trees and low branches) within the forest canopy (Ryan et al., 2013). Numerous studies have shown prescribed fire to reduce subsequent wildfire behavior (Finney et al., 2005; Prichard et al., 2010; Cochrane et al., 2012) (but see Fernandes and Botelho, 2003). However, there are relatively few long-term data documenting the duration of prescribed fire effectiveness in reducing fire hazard (Stephens et al., 2009; Vaillant et al., 2009), and many studies have been confined to a particular forest type or specific locale (Battaglia et al., 2008; Safford et al., 2012; Stephens et al., 2012). The effectiveness of prescribed fire may be obscured by many potential sources of variability, including fuel structure and condition, climate, forest type, and fire weather and ignition patterns (e.g., strip head fires versus aerial ignitions, which may create more intense fires) (Agee and Skinner, 2005; Ryan et al., 2013).

While prescribed fire is expected to reduce surface fuels, it is possible that post-fire recruitment of woody fuels may be high from branch abscission and tree fall from fire-caused mortality, perhaps surpassing pre-fire woody fuel amounts (Agee and Skinner, 2005; Keifer et al., 2006). Multiple prescribed fires may be required before the desired long-term fuel reductions are achieved (North et al., 2007; Ritchie et al., 2007). Yet there has been little work to quantify how many fires, and at what interval, may be needed to achieve substantial long-term fuels reductions.

* Corresponding author.
E-mail addresses: pvanmantgem@usgs.gov (P.J. van Mantgem), llalemand@usgs.gov (L.B. Lalemand), MaryBeth_Keifer@nps.gov (M. Keifer), Jeffrey.Kane@humboldt.edu (J.M. Kane).

http://dx.doi.org/10.1016/j.foreco.2016.07.028
0378-1127/Published by Elsevier B.V.

From a fire hazard perspective, the relevant measures of success of prescribed fire is arguably not changes in fuel loads, but rather the magnitude and duration of the inhibition of expected fire behavior. Estimating these effects is perhaps best addressed using models of fire behavior (e.g., Rebain et al., 2009). While these models may underrepresent actual fire hazards (Cruz and Alexander, 2010), a consistent framework allows the assessment of relative changes across locations and over time. Using this approach, we used long-term observations of prescribed fire effects in coniferous forests of U.S. national parks in California and the Colorado Plateau to address several basic forest fuels management questions, (1) how effective is prescribed fire at reducing fuels and moderating estimates of subsequent fire behavior? (2) How long are treatments effective at reducing fuels and estimates of fire hazards? (3) What differences emerge from first- and second-entry prescribed fires? and (4) how do prescribed fire effects vary across broad forest types and geographic regions? Prescribed fire represents a substantial investment of resources, so critically assessing treatment effectiveness is a crucial step towards adaptive management of western forests.

2. Methods

2.1. Data sources

We capitalized on existing plot-based prescribed fire monitoring data from the National Park Service's (NPS) fire ecology program, using the interagency FFI (FEAT/FIREMON Integrated) database management system (https://www.frames.gov/partner-sites/ffi/ffi-home/). Beginning in the early 1990s, the NPS developed a standardized monitoring protocol, which allows direct comparisons of fire effects between and within prescribed fire units, regions and years. Standard NPS protocols establish at least one 50×20 m plot (0.1 ha) at a random location within the prescribed fire unit prior to prescribed fire (NPS, 2003). Within each plot four permanently marked 15.24 m fuel transects are established (Brown et al., 1982). Fuels are sampled with varying intensity by size class, with 0–0.62 cm diameter fuels (1 h) and 0.62–2.54 cm diameter fuels (10 h) tallied from 0 to 1.83 m along the transect, 2.54–7.62 cm diameter fuels (100 h) tallied from 0 to 3.66 m along the transect, and >7.62 cm diameter fuels (1000 h) tallied along the entire transect length. Litter and duff depths are measured with 5–10 samples per transect. Using the fuels transect data we estimated wood debris loading, and converted litter and duff depth to weights using bulk density values (Brown, 1974; Brown et al., 1982). At each plot survey trees are measured for diameter at breast height (DBH, 1.37 m) and live/dead status. Within each plot all overstory trees (>15 cm DBH) are measured, while smaller trees ($\leqslant$15 cm DBH, with a minimum DBH generally >2.5 cm) are measured within subplots, typically over an area of 25×10 m. The plots were surveyed pre-fire and at multiple intervals post-fire. Typical post-fire monitoring intervals were immediately post-fire, 1-, 2-, and 5-years post-fire, and at subsequent 5- to 10-year intervals.

2.2. Data quality assurance

The range of each numeric field in our dataset was checked to ensure all values fell within an appropriate range and that measurements were taken on a consistent scale (e.g., cm vs. m). Other errors that were identified included trees with multiple species listed on separate records, multiple observations of the same tree recorded on the same day, illogical changes in DBH over repeated measurements and inconsistent measures of tree status through time (i.e., trees listed as live after being listed as dead). Monitoring protocols consistently recorded the year of field data collection, but often omitted the year of prescribed fire ignition. Where this occurred, we inferred the calendar year of fire from field notes and the year of data collection. On three occasions, we corrected single observations of either fine or coarse fuels measurements that were an order of magnitude larger than their particular time series. A single plot in Sequoia National Park appeared to be an outlier in terms of fuel amounts and was excluded from all analyses. We obtained similar results if this plot was included.

We only considered plots that had complete observations for pre- and post-fire fuels, had at least three post-fire measurements, were dominated by conifers, and were estimated to have a timber or slash fuel model type (see Section 2.3). To determine longer-term patterns, we restricted analyses to plots where the final observation was taken >7 years post-fire for first-entry burns, and >4 years post-fire for second-entry burns (first-entry = 117 plots, second-entry = 36 plots). Tests of changes in stand density and basal area had the additional restriction that understory trees (<15 cm DBH) were measured at each observation interval (first-entry = 55 plots, second-entry = 9 plots). We divided the plots into groups by geographic region (California and Colorado Plateau), and forest community type defined by plot basal area dominants; mixed conifer forest, primarily dominated by fir (*Abies* spp.), and ponderosa pine forest, dominated by *Pinus ponderosa*. We had sufficient data for separate analyses of three broad forest types: California mixed conifer (57 plots), California ponderosa pine (33 plots), and Colorado Plateau ponderosa pine (27 plots). Other regions and forest types were too sparse to consider. Our data set ranged across nine national parks in four western states (Arizona, California, New Mexico, and Utah), spanning 6° latitude and 16° longitude (Table 1). Second-entry prescribed fires occurred on average 10 years following the first-entry fires (range of years elapsed between fires: 6–17 years). For second-entry fires we assessed changes compared to conditions immediately before second-entry prescribed fire (i.e., not relative to the original first-entry pre-fire conditions).

2.3. Fire behavior estimation

We estimated fire behavior for each plot at each measurement using individual records of fuels and forest structure in combination with information on local fire weather (i.e., we did not simulate future conditions). Fire behavior was estimated for each plot survey interval using the Fire and Fuels Extension to the Forest Vegetation Simulator (FFE-FVS) fire simulation model (Rebain et al., 2009), using appropriate geographic variants for FFE-FVS for each park unit. The observed fuels data were used by FFE-FVS algorithms to select the closest standard fuel model (Anderson, 1982), which provides an objective and well-documented method to select fuel models. FFE-FVS uses observed fuel loads and forest stand conditions (e.g., dominant tree species and canopy cover) to estimate a standard fuel model, though the specifics of how this information is used differs among geographic variants of the model. Canopy fuels, including stand canopy base height (m) and canopy bulk density (kg m^{-3}) are estimated by FFE-FVS following Scott and Reinhardt (2001), using allometric relationships between forest inventory data (primarily tree diameter) to estimate canopy characteristics. FFE-FVS follows Rothermel (1972), using Byram's equations (Alexander and Cruz, 2012) to estimate flame length. To highlight potential hazards, we report FFE-FVS estimates of total flame length under severe weather conditions, taking crown fire activity into account. We removed plots from analyses of crown fire hazard where understory trees were not measured consistently or where FFE-FVS determined canopy fuels were too sparse to estimate (first-entry = 97 plots, second-entry = 29 plots). We emphasize that we did not use FFE-FVS to project future fire behavior,

Table 1
Summary information for fire monitoring plots across national park units.

Park[a]	Lat. (°)	Long. (°)	Plot count	Average density (trees ha^{-1})	Average basal area (m^2 ha^{-1})	Modal pre-fire fuel model		Dominant species by plot[b]
						First-entry	Second-entry	
BAND	35.8	−106.3	2	125	0.3	10		PIPO 100%
BRCA	37.6	−112.2	5	856	17.5	8	10	ABCO 44%, PIPO 38%, PSME 15%, JUSC 2%, PIFL 2%
GRCA	36.2	−112.7	10	1133	41.3	10	10	PIPO 78%, ABCO 15%, POTR 3%, JUOS 1%, PIED 1%, PIEN 1%, PSME 1%
LABE	41.8	−121.5	4	309	22.3	9		PIPO 100%
LAVO	40.5	−121.4	25	708	48.3	10	10	ABCO 47%, PIPO 38%, PIJE 7%, CADE 6%, PICO 1%
SEKI	36.5	−118.6	45	658	129.5	10	10	SEGI 56%, ABCO 24%, PILA 10%, CADE 4%, ABMA 3%, PIPO 2%, QUKE 1%
WHIS	40.6	−122.6	7	484	45.2	9	12	ABCO 41%, PILA 38%, PIPO 20%, QUKE 1%
YOSE	37.9	−119.6	9	412	45.3	10	10	ABCO 50%, CADE 18%, PILA 12%, PIPO 9%, PIJE 7%, QUKE 3%
ZION	37.3	−113.0	10	1213	31.0	8	8	PIPO 79%, ABCO 12%, JUSC 5%, QUGA 3%

[a] National park unit acronyms refer to the following: BAND = Bandelier NM, BRCA = Bryce Canyon NP, GRCA = Grand Canyon NP, LABE = Lava Beds NM, LAVO = Lassen Volcanic NP, SEKI = Sequoia and Kings Canyon NP, WHIS = Whiskeytown NRA, YOSE = Yosemite NP, ZION = Zion NP.
[b] Pre-fire plot dominant species by basal area. Percentages may not add to 100 due to rounding. ABCO = *Abies concolor*, ABMA = *Abies magnifica*, CADE = *Calocedrus decurrens*, JUOS = *Juniperus osteosperma*, JUSC = *Juniperus scopulorum*, PICO = *Pinus contorta*, PIED = *Pinus edulis*, PIEN = *Picea engelmannii*, PIFL = *Pinus flexilis*, PIJE = *Pinus jeffreyi*, PILA = *Pinus lambertiana*, PIPO = *Pinus ponderosa*, POTR = *Populus tremuloides*, PSME = *Pseudotsuga menziesii*, QUGA = *Quercus gambelii*, QUKE = *Quercus kelloggii*, SEGI = *Sequoiadendron giganteum*.

rather FFE-FVS outputs were obtained using only the observed forest inventory data at each plot measurement.

Potential fire weather data were derived from nearby Remote Automated Weather Stations (RAWS) that best represented individual plot locations and had at least 10 years of continuous records. We used quality controlled RAWS daily data available through the National Fire and Aviation Management web applications (http://fam.nwcg.gov/fam-web/weatherfirecd/state_data.htm, data last accessed March 2013), supplemented with quality checked hourly RAWS data from the Western Regional Climate Center (http://www.raws.dri.edu/, data last accessed March 2013). Fire Family Plus (Bradshaw and McCormick, 2000) was used to generate daily averages of fire weather data for each location during peak fire seasons. We filtered the daily weather data to include only measurements recorded during the fire season, with months of the peak fire season determined by ecoprovince of each NPS unit following Littell et al. (2009). We calculated the 97.5th percentile fire weather for each of the RAWS station data (Table S1), which was used to parameterize the FFE-FVS fire behavior model. We also calculated and analyzed FFE-FVS estimates using 80th and 90th percentile fire weather, but these conditions did not give us appreciably different results.

2.4. Data analysis

For each forest type we assessed differences between pre-fire and immediate post-fire measurements, and differences between pre-fire and final post-fire measurements for stand density (trees ha^{-1}), basal area (m^2 ha^{-1}), fuels by type [duff, litter, fine woody (0–7.62 cm diameter, 1–100–h), and coarse woody fuels (>7.62 cm diameter, 1000-h), in Mg ha^{-1}], canopy base height (m), canopy bulk density (kg m^{-3}), and estimated flame lengths (m) using paired permutation tests. For each set of comparisons (e.g., changes in first-entry fuels), we used *P*-values adjusted by the false discovery rate, the expected proportion of type I errors among the rejected null hypotheses (similar to, but more powerful, than the familiar Bonferroni correction) (Benjamini and Hochberg, 1995).

Where the paired permutation tests suggested evidence of a return to pre-fire conditions, we used linear mixed-effects models (LMM) to identify time-dependent changes (Gelman and Hill, 2007). We estimated LMMs using maximum likelihood, with the plot identifier modeled as random intercepts and slopes. We determined model support using the Akaike Information Criterion, corrected for sample size (*AICc*) (Burnham and Anderson, 2002). We used differences in *AICc* of >4 to identify models with different levels of support. Strong spatial autocorrelation of fire effects among plots within a burn unit may be uncommon (van Mantgem and Schwilk, 2009), and we considered the plot as our sampling unit. We tested improvements to our models by nesting plots within burn units and within park units, but these grouping strategies were generally not supported using *AICc* (and when applied, nesting plots within burn units did not qualitatively change model results). The LMMs allowed us to account for variable plot census intervals (e.g., time-series data based on year since burn, t) while incorporating plot-specific variance. The LMMs can be written as

$$Y_{ij} = (\beta_1 + b_{1i}) + (\beta_2 + b_{2i})t_{ij} + \varepsilon_{ij},$$

$$\text{where } b_{1i} \sim N(0, \sigma_1^2),\ b_{2i} \sim N(0, \sigma_2^2),\ \text{and } \varepsilon_{ij} = \phi\varepsilon_{i,j-1} + \alpha_{ij},$$

b_{1i} and b_{2i} are the random intercept and random slope terms, and $\boldsymbol{b}_i = (b_{1i}, b_{2i})'$ are assumed to be independent. Error was modeled as normally distributed independent noise (α_{ij}) plus within plot temporal autocorrelation using a first order autocorrelation function, $h(k, \phi) = \phi^k$, $k = 0, 1, \ldots$ where k is the years between observations. To evaluate patterns among plots we considered absolute changes compared to conditions immediately prior to the prescribed fire. Qualitatively similar results were obtained using relative differences from pre-fire conditions. Linear model assumptions were checked using residual plots. Because LMMs directly model the effects of among plot variance, they allow for an interpretation of average trends.

We determined improvements of LMMs by the addition of categorical variables for estimated pre-fire fuel model. We had sufficient data to model differences in fuel models 8 (compact timber litter), 10 (timber understory) and 12 (medium slash) for California mixed conifer plots, and fuel models 8 and 10 for California and Colorado Plateau ponderosa pine plots. All analyses were conducted using the R statistical language (R Development Core Team, 2015), with LMMs created using the 'nmle' package. The proportion of variation explained using the fixed effects only (marginal R^2) and the combined fixed and random (grouping) effects terms (conditional R^2) of the fitted models were calculated following Nakagawa and Schielzeth (2013) using the 'MuMIn' package.

3. Results

Stand density was reduced immediately following first-entry fires across all forest types (paired permutation tests, $P \leqslant 0.02$), and remained reduced >7 years post-fire (paired permutation tests, $P \leqslant 0.01$) (Fig. 1). Immediate post-fire reductions in basal area were also significant (paired permutation tests, $P \leqslant 0.01$), but only remained significantly reduced for California mixed conifer plots by >7 years post-fire (paired permutation test, $P \leqslant 0.01$). Combining all regions and forest types, prescribed fire led to greater immediate reductions in stand density (31% median reduction) compared to basal area (average 13% reduction). This pattern held by the final observations (stand density = 51% median reduction; basal area = 8% median reduction).

Surface fuels were reduced following prescribed fire across all forest types, and plot-level estimates of fuel models generally changed following prescribed fire (Table 2). Patterns of response to prescribed fire varied by fuel particle size classes (Fig. 2). Both duff and litter fuels across all forest types were immediately reduced post-fire (paired permutation tests, $P < 0.01$), and remained below pre-fire levels by >7 years post-fire (paired permutation tests, $P < 0.02$). Across most forest types both fine and coarse fuels were reduced immediately following prescribed fire (paired permutation tests, $P < 0.01$), with the exception for fine fuels in Colorado Plateau ponderosa pine plots where the reduction was marginal (paired permutation test, $P = 0.07$). Fine and coarse fuels showed evidence of returning to pre-fire levels by >7 years post-fire for all forest types (paired permutation tests, $P \geqslant 0.35$).

Most estimates of potential fire hazards showed significant reductions immediately post-fire (paired permutation tests, $P < 0.04$) (Fig. 3), with the exception of estimated canopy height in California and Colorado Plateau ponderosa pine plots (paired permutation tests, $P \geqslant 0.14$) and flame length in Colorado Plateau pine plots (paired permutation tests, $P = 0.59$). We found little evidence that measures of potential fire hazard were similar to pre-fire conditions by >7 years post-fire (paired permutation tests, $P \leqslant 0.04$), excluding measures where we did not observe significant changes immediately post-fire.

Table 2
Modal estimated fuel model for pre- and post-fire intervals by region and forest type.

Prescribed fire type	Region	Forest type	Pre-fire	Year post-fire		
				0–3	4–7	>7
First-entry	California	Mixed conifer	10	8	10	10
		Ponderosa pine	10	9	9	10
	Colorado Plateau	Ponderosa pine	8	8	8	8
Second-entry	California	Mixed conifer	10	8	10	
		Ponderosa pine	10	8	8	
	Colorado Plateau	Ponderosa pine	8	8	8	

Second-entry prescribed fires appeared to reduce stand density, basal area, and estimates of fire hazards, but these reductions were not statistically significant (paired permutation tests, $P \geqslant 0.21$) (Figs. S1 and S3). Second-entry fire also appeared to reduce surface fuels, but these results were often non-significant, or marginally significant (paired permutation tests, $P = 0.05$–0.81) (Fig. S2). Exceptions were duff and fine fuels in California mixed conifer plots, and litter fuels in Colorado Plateau ponderosa pine plots (paired permutation tests, $P \leqslant 0.03$). Of these, duff and fine fuels in California mixed conifer plots suggested returns to pre-fire conditions by >4 years post-fire (paired permutation tests, $P \geqslant 0.15$). Note that our sample size was much lower and our sampling duration was shorter for second-entry fires compared to first-entry fires.

Mixed-effects models, which describe general time-dependent changes among plots, support the idea that immediate fine and coarse fuels reductions were generally substantial, and that in return times to pre-fire conditions for fine and coarse fuels were often more than a decade (Tables 3 and 4). There appeared to be differences among pre-fire fuel models by *AICc* (Table S2). For ponderosa pine plots, we did not find differences in rates of return between plots located in California and the Colorado Plateau (mod-

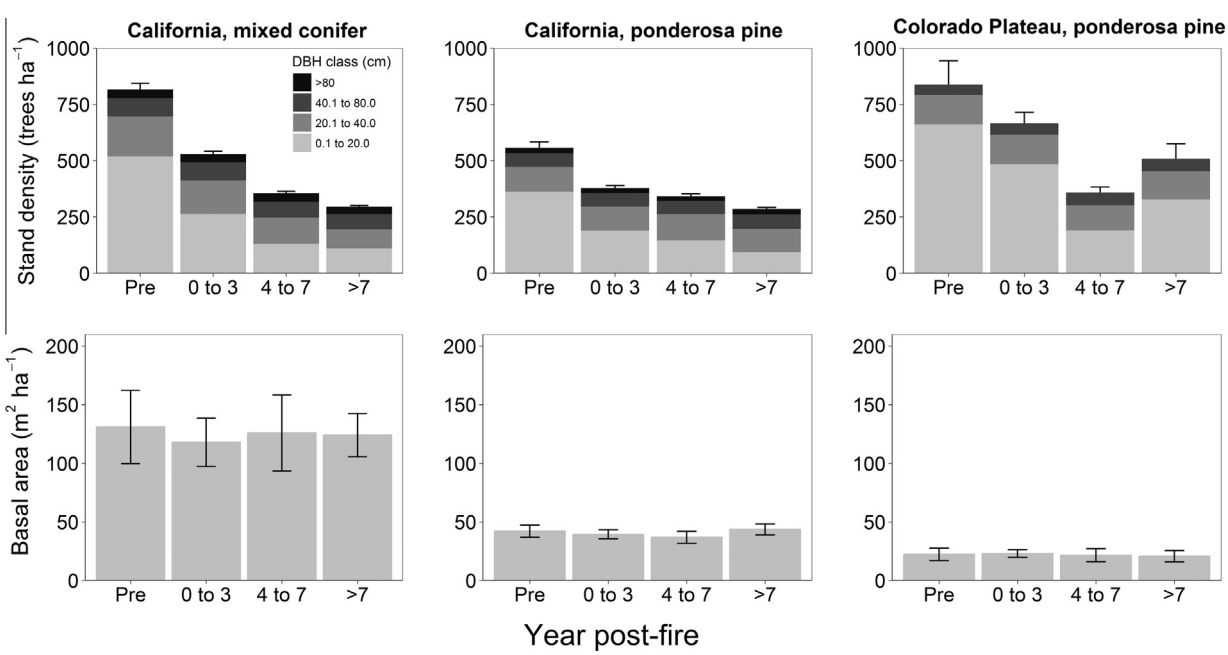

Fig. 1. Average stand density and basal area by forest type before and after first-entry prescribed fire. Error bars represent one standard error of the mean. High basal area values for California mixed conifer stands were associated with the presence of large giant sequoia.

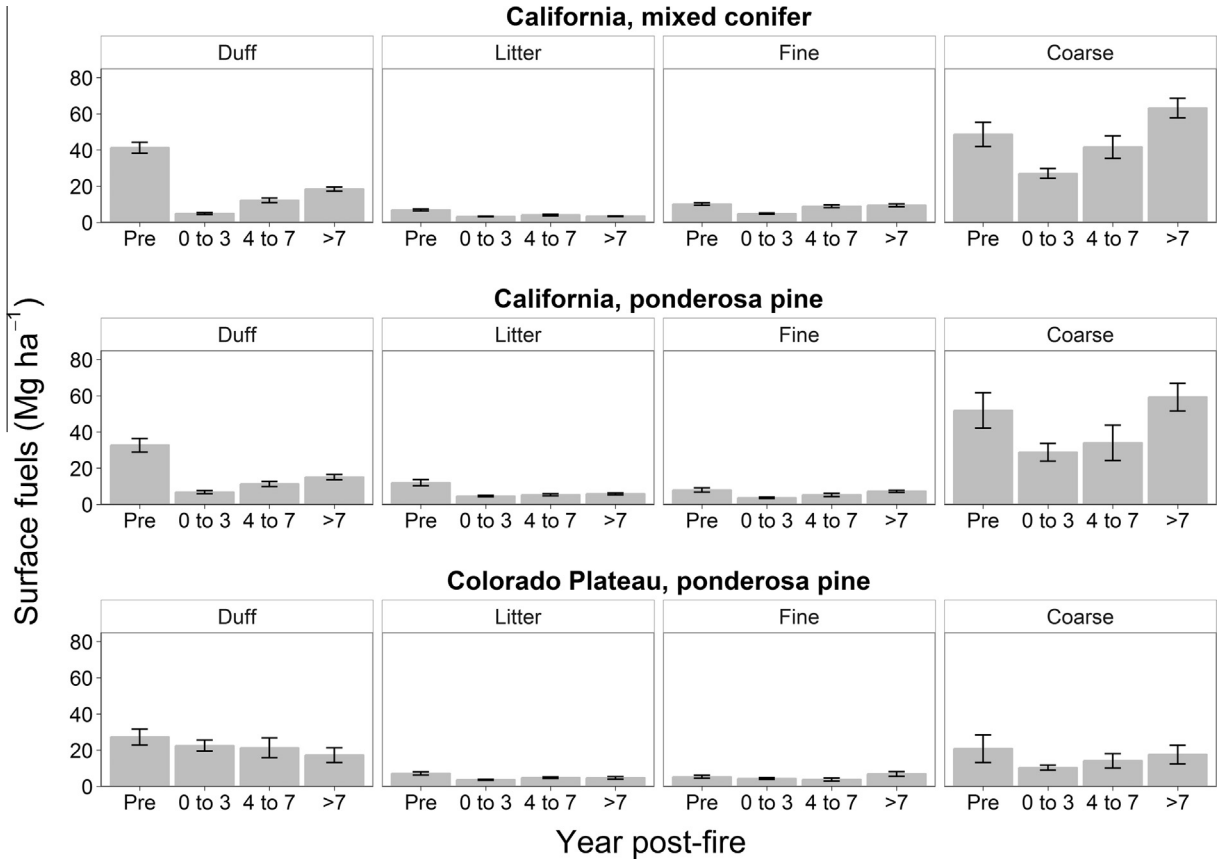

Fig. 2. Average surface fuels loading by forest type before and after first-entry prescribed fire. Error bars represent one standard error of the mean.

els with an additional term for regional differences did not have substantially lower *AICc* values). Commonly, where pre-fire fuel models suggested heavier surface fuels the initial reductions from prescribed fire were more substantial and showed slower rates of return to pre-fire conditions. The models for first-entry prescribed fires explained a majority of the variation (first-entry, marginal $R^2 \geqslant 0.10$, conditional $R^2 \geqslant 0.51$). The relatively low values for the marginal R^2 versus conditional R^2 highlight the large among-plot variation. We were unable to create acceptable models for time-dependent changes in surface fuels following second-entry fires (95% CIs for β_{Year} overlapped zero).

4. Discussion

Our analyses show that prescribed fire leads to an immediate reduction in fuels and fire hazard and, depending on the measure of fuel type or fire hazard, reductions may last a decade or longer in some western coniferous forests. Our findings of immediate reduction of hazard following prescribed fire are in general agreement with earlier studies conducted in western forests (Stephens et al., 2009, 2012; Vaillant et al., 2009; Noonan-Wright et al., 2014). The duration of fire hazard reduction is also in agreement with other studies, such as in the coniferous forests of California, where Stephens et al. (2012) and Vaillant et al. (2015) showed that fuels and potential fire behavior were reduced at least to seven years following prescribed fire, and Lydersen et al. (2014) found that areas that experienced fire within 14 years typically re-burned at low severity (unless wildfire weather was extreme).

Different measures of prescribed fire effectiveness, including reductions in fuel loads and potential fire behavior, showed varying degrees of immediate post-fire reductions and rates of return to pre-fire conditions. Duff and litter surface fuels did not appear to return to pre-fire levels over the course of our observations. Many of these sites had likely not burned for over a century (e.g., Swetnam et al., 2009), so it could take decades for stands to develop characteristics similar to what was observed prior to prescribed burning. Fine and coarse fuels did appear to return to pre-fire levels, and were likely recruited from branches and boles of trees killed in the prescribed fire, perhaps allowing a faster return to pre-fire fuel loads. Measures of crown fire hazard (estimated canopy base height, crown bulk density, and flame lengths) did not suggest a return to pre-fire conditions over the duration of our observations. This result may be because fire reduced tree density to a greater degree than stand basal area, suggesting prescribed fire disrupted vertical fuel continuity by removing small trees (van Mantgem et al., 2011, 2013). Thus, crown fire hazards may be reduced until stands develop a new cohort of small trees.

There is little available published work documenting the effects of repeated prescribed fires (Webster and Halpern, 2010), and this information gap has been identified as an important research need (Stephens et al., 2012). While we had a limited number of second-entry sites, our data suggest that second-entry prescribed fires reinforced changes following first-entry fires, reducing surface fuels that had built up since the first-entry fires (though reductions were mostly not statistically significant). Webster and Halpern (2010) found similar results in terms of second-entry burning on surface fuels at Sequoia and Kings Canyon National Parks, though the present study likely uses some of the same plot data. Compared to first-entry fires, second-entry fires appeared to have a more muted impact on stand density, stand basal area, and estimates of crown fire hazard (estimated canopy base height, crown bulk density, and flame length), likely as surviving tree crowns and

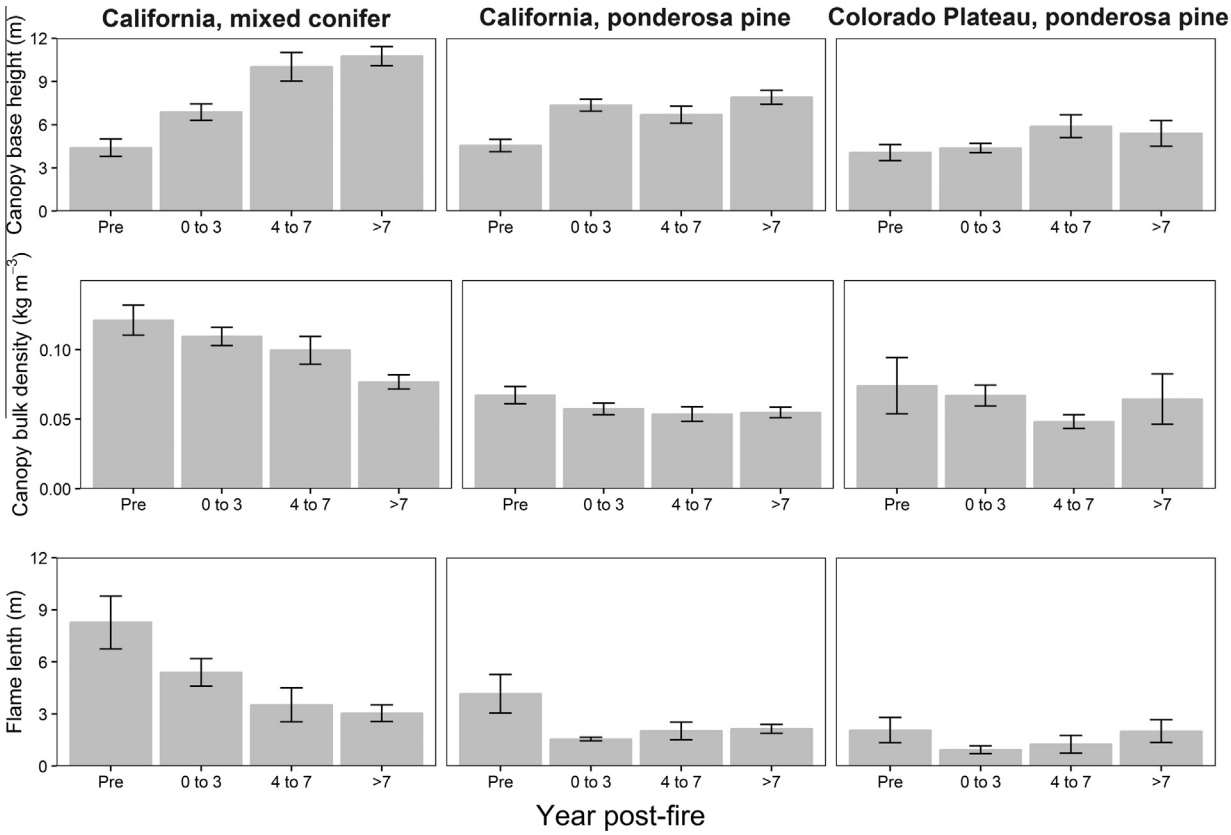

Fig. 3. Average estimated canopy structure and flame lengths under 97.5% fire weather conditions by forest type before and after first-entry fires. Error bars represent one standard error of the mean.

Table 3
Linear mixed model averaged estimates for time-dependent changes in fine and coarse surface fuels following first-entry fires. Comparisons among pre-fire fuel models are presented relative to fuel model 8. Note, the data did not suggest time-dependent changes in duff and litter fuels, and for any fuel size class following second-entry fires. The 95% CIs represent uncertainty in the fixed effects only, not among plot variability.

Forest type	Fuel type	Parameter	Average estimate (95% CI)	Model marginal R^2	Model conditional R^2
Mixed conifer	Fine	β_0	−2.95 (−5.83 to −0.07)	0.175	0.512
		β_{year}	0.41 (0.26–0.56)		
		β_{FM10}	−2.27 (−5.97 to 0.43)		
		β_{FM12}	−3.73 (−7.89 to −1.21)		
	Coarse	β_0	−5.4 (−18.32 to 7.52)	0.181	0.591
		β_{year}	3.00 (1.53–4.47)		
		β_{FM10}	−15.44 (−33.07 to 2.18)		
		β_{FM12}	−30.71 (−49.05 to −12.36)		
		$\beta_{year*FM10}$	−0.11 (−3.89 to 2.12)		
		$\beta_{year*FM12}$	−0.14 (−4.24 to 2.03)		
Ponderosa pine	Fine	β_0	−2.8 (−4.25 to −1.34)	0.100	0.660
		β_{year}	0.38 (0.15–0.60)		
		β_{FM10}	0.02 (−2.26 to 2.34)		
		$\beta_{year*FM10}$	−0.10 (−0.52 to 0.00)		
	Coarse	β_0	3.57 (−8.25 to 15.4)	0.212	0.598
		β_{year}	2.8 (0.96 to 4.64)		
		β_{FM10}	−33.4 (−49.83 to −16.97)		
		$\beta_{year*FM10}$	0.01 (−3.16 to 3.23)		

small trees did not have time to develop by the time of second-entry prescribed fires.

Patterns between forest types (mixed conifer and ponderosa pine stands) and biogeographic area (California versus Colorado Plateau) suggested only moderate differences in how these forest types responded to prescribed fire. Models comparing fine and coarse fuel accumulation in mesic mixed conifer versus more arid ponderosa pine forest types suggested more arid sites had lower pre-fire fuel loading, slower rates of post-fire fuel accumulation and showed less absolute change in crown fire hazards following prescribed fire, however confidence interval estimates between these forests types overlapped for estimates of initial fuel consumption and rate of return to pre-fire conditions. Forest type helped explain some of the variation in our data, but these categories do not provide a mechanistic understanding of the forces that were responsible for the large amount of among-plot variability in prescribed fire response. Likely factors that may help better predict individual prescribed fire responses include ignition

Table 4
Linear mixed model predictions for average initial post-fire change and time-dependent changes in fine and coarse surface fuels following first-entry fires. The maximum post-fire year observed was 22 years following first-entry fires. The 95% CIs represent uncertainty in the fixed effects only, not among plot variability.

Forest type	Fuel type	Pre-fire fuel model	Average initial post-fire change Mg ha^{-1} (95% CI)	Average return time to pre-fire conditions years (95% CI)
Mixed conifer	Fine	8	−2.5 (−3.7 to −1.3)	7 (4–10)
		10	−5.3 (−6.4 to −4.2)	13 (10–17)
		12	−7.1 (−8.3 to −5.9)	18 (14–22)
	Coarse	8	−5.2 (−11.7 to 1.2)	2 (0–5)
		10	−20.9 (−26.8 to −15)	8 (6–10)
		12	−36.2 (−42.6 to −29.8)	13 (10–17)
Ponderosa pine	Fine	8	−2.8 (−3.7 to −2)	6 (5–9)
		10	−2.8 (−3.5 to −2)	13 (9–20)
	Coarse	8	3.6 (−2.4 to 9.5)	0 (0–1)
		10	−29.8 (−35.4 to −24.3)	11 (8–15)

fire weather, ignition pattern, site topographic position, average local climate (particularly estimates of the water balance), and stand species composition (Kane et al., 2015; Lydersen et al., 2015).

While our observations provide an exceptional data set to explore how fire hazards change following prescribed fire, our results have several limitations. First, our data represent fire effects only following prescribed fire. Wildfires may lead to more pronounced reductions in subsequent fire hazards and may be part of a fuels management strategy (i.e., management of wildland fire) (Collins and Stephens, 2007), but may also cause undesired changes in forest structure, such as the death of large trees. Second, our data did not include the effects of combined mechanical thinning and prescribed fire treatments, which are common in some forest types. Mechanical thinning followed by prescribed fire has been shown to reduce some measurements of fire hazards (e.g., canopy bulk density) to a greater degree than prescribed fire alone (North et al., 2007; Schmidt et al., 2008; Knapp et al., 2012; Stephens et al., 2012). How forests treated with mechanical thinning and prescribed fire respond to repeated prescribed fire has not been well documented. Third, we considered forest fuel types only. We expect forests with grass fuel types to respond differently to prescribed fire, perhaps with surface and live fuels returning relatively quickly to pre-fire conditions. Last, we used models to estimate some dimensions of fire hazard. These models make simplifying assumptions and may underestimate fire behavior (Cruz and Alexander, 2010). We emphasize, however, that we were interested in estimating changes in pre- versus post-fire hazards, not deriving precise estimates of fire behavior.

Reducing fire hazard is important, but forests are managed in the context of several competing concerns, including maintenance of biodiversity, watershed integrity and, increasingly, carbon sequestration (Campbell et al., 2011; Hurteau and Brooks, 2011). Prescribed fire appeared to mostly remove small trees, so that carbon losses from live vegetation might be moderate. Increased growth of residual trees following fire may also help offset carbon losses. It is possible that overall reductions in stand-level carbon from prescribed fire may be modest in the dry coniferous forest types we considered, although additional work is needed to describe these patterns.

Our results suggest that prescribed fire can be an important tool to reduce fire hazards, perhaps helping to create stands that are more resistant (sensu Walker et al., 2004) to expected higher frequencies of extreme fire weather (Moritz et al., 2012). Thinning forest stands via prescribed fire may also reduce competition among trees, perhaps promoting resistance and resilience to other forest stressors such as drought (D'Amato et al., 2013; van Mantgem et al., 2016). How well prescribed fire allows managers to meet these challenges remains to be seen, and represents a critical avenue of future research. Continuing, and perhaps expanding, current fire effects monitoring efforts will be essential for this work.

Acknowledgements

We thank the many scientists and field crews who collected and organized the fire effects data. Jon Hollis assisted in the organization of the data. Morgan Varner and two anonymous reviewers provided helpful comments on an earlier version of this manuscript. Julie Yee provided statistical advice. This project was partially supported by the Joint Fire Science Program (JFSP project number 12-1-03-25). Any use of trade, firm or product names is for descriptive purposes only and does not imply endorsement by the U.S. Government.

Appendix A. Supplementary material

Supplementary data associated with this article can be found, in the online version, at http://dx.doi.org/10.1016/j.foreco.2016.07.028.

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