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Long-term effects of prescribed fire on mixed conifer forest structure in the Sierra Nevada, California

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ABSTRACT

The capacity of prescribed fire to restore forest conditions is often judged by changes in forest structure within a few years following burning. However, prescribed fire might have longer-term effects on forest structure, potentially changing treatment assessments. We examined annual changes in forest structure in five 1 ha old-growth plots immediately before prescribed fire and up to eight years after fire at Sequoia National Park, California. Fire-induced declines in stem density (67% average decrease at eight years post-fire) were nonlinear, taking up to eight years to reach a presumed asymptote. Declines in live stem biomass were also nonlinear, but smaller in magnitude (32% average decrease at eight years post-fire) as most large trees survived the fires. The preferential survival of large trees following fire resulted in significant shifts in stem diameter distributions. Mortality rates remained significantly above background rates up to six years after the fires. Prescribed fire did not have a large influence on the representation of dominant species. Fire-caused mortality appeared to be spatially random, and therefore did not generally alter heterogeneous tree spatial patterns. Our results suggest that prescribed fire can bring about substantial changes to forest structure in old-growth mixed conifer forests in the Sierra Nevada, but that long-term observations are needed to fully describe some measures of fire effects.

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1. Introduction

Fire exclusion in many forests across the western U.S. has led to changes in forest structure, such as high surface fuel loads, high densities of small stems that act as ladder fuels to promote crown fires, and increasing dominance of shade-tolerant species. These changes are particularly acute in forests that historically had low severity/high frequency fire regimes (Allen et al., 2002; Brown et al., 2004; Agee and Skinner, 2005; Noss et al., 2006). In response to high fuel accumulations, managers have used prescribed fire to reduce surface fuels and small tree density, particularly for shade-tolerant species, while preserving large trees (i.e., individuals presumed to have established prior to Euro-American settlement, ca. 1850). Changes to forest structure following prescription burning should also reduce risks for stand replacing fires while maintaining the majority of above-ground live tree carbon stocks (van Wagtendonk, 1996; Stephens and Moghaddas, 2005; Hurteau and North, 2009; North et al., 2009).

How effective is prescribed fire at restoring forest structure? Large-scale trials have shown that prescription burning results in reduced fuel loads and stem density, but that a single application of prescription burning (even in combination with mechanical thinning) is unlikely to recreate presumed historic forest structure (North et al., 2007; Schwilk et al., 2009). Judgments of prescribed burning as a restoration tool might be improved by considering the long-term effects of fire. Studies generally measure the effects of prescribed fire only a few years following fire (Hood, 2010), although delayed mortality of trees can occur over longer intervals, as pathogens or other stressors kill fire-damaged trees (Schwilk et al., 2006, 2009; Hood and Bentz, 2007; Hood, 2010) (but see Youngblood et al., 2009). Although the bulk of fire-caused tree mortality occurs immediately after burning, it is unclear how long mortality rates remain elevated following burning, and if these delayed effects constitute an important influence on post-fire forest structure.

An additional concern, at least in some forest types of the Sierra Nevada of California, is that fire exclusion is thought to have altered the spatial arrangement of trees within stands. The spatial arrangement of stems (the degree to which the distribution of trees in a stand can be considered clumped, random or uniform) is an important structural element of stands, in that it defines local competitive

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Table 1
Characteristics of forest monitoring plots.

Plot identifier	Lat. (°N)	Long. (°W)	Elevation (m)	Plot area (ha)	Establishment year	Species comprising >1% of stems[a]
FFS2	36.6	−118.7	2128	1.00	2001	ABCO 65%; ABMA 26%; CADE 1%; PILA 8%
FFS5	36.6	−118.8	2030	1.00	2001	ABCO 57%; CADE 25%; PILA 18%
FFS6	36.6	−118.8	2018	1.00	2001	ABCO 62%; CADE 3%; PIJE 1%; PILA 24%; PIPO 1%; QUKE 10%
LOTHAR	36.6	−118.7	2167	1.13	1984	ABCO 74%; ABMA 1%; PIJE 1%; PILA 16%; QUKE 6%; SEGI 2%
UPTHAR	36.6	−118.7	2202	1.00	1984	ABCO 95%; PIJE 2%; PILA 2%

[a] Species composition of stems at time of plot establishment. Percentages may not add to 100 due to rounding. ABCO = *Abies concolor*, ABMA = *A. magnifica*, CADE = *Calocedrus decurrens*, PIJE = *Pinus jeffreyi*, PILA = *P. lambertiana*, PIPO = *P. ponderosa*, QUKE = *Quercus kelloggii*, SEGI = *Sequoiadendron giganteum*. Naming conventions follow Hickman (1993).

environments, which in tern influences individual tree growth rates (Biging and Dobbertin, 1992) and mortality risks (Das et al., 2008). Within the Sierran mixed conifer, fire exclusion has resulted in high densities of small trees, particularly of shade-tolerant *Abies concolor* (white fir), a development that can result in an increase in clumping of the spatial pattern of stems (North et al., 2007). The reduction of stem clumping, particularly for small trees, has been identified as a restoration goal in the Sierra Nevada (Taylor, 2004; North et al., 2007).

Here we examine if prescribed fire results in forest structure that is consistent with common restoration targets. We present changes to forest structure as measured by eight years of annual monitoring at five stands (1 ha) burned in four separate fires in old-growth mixed conifer forests in the Sierra Nevada. Our study is notable in that we consider multiple fires separated both in space and time, and can account for long-term delayed tree mortality. Specifically, we describe the effects of prescribed fire on stand density, live stem biomass, stem size distributions, species composition, and spatial arrangement of stems.

2. Materials and methods

2.1. Study sites

We compared pre- and post-fire forest structure for all living trees greater than 1.37 m in height in five 1 ha plots of old-growth mixed conifer forest in the Giant Forest area, Sequoia National Park (Table 1). Prior to prescription fires, the Giant Forest area last burned in the 1860s or 1870s (Schwilk et al., 2006; Swetnam et al., 2009). The climate of this area is Mediterranean, with wet, snowy winters and long, dry summers. The mean annual precipitation of the Giant Forest area is 1255 mm at an elevation of 1950 m, with approximately half of this precipitation arriving as snow (Stephenson, 1988). Average January and July air temperatures are 0 °C and 18 °C, respectively. The plots lie at a minimum of 300 m and a maximum of 3700 m apart and are of comparable size, shape, average slope steepness, average slope aspect, and soil parent material (granodiorite) (Table 1). Prior to burning the plots were dominated by *A. concolor* (average of 70% of stems), but with considerable representation of *Pinus lambertiana* (average of 14% of stems). Further details on the plots can be found in Knapp and Keeley (2006), Knapp et al. (2007), and Mutch and Parsons (1998).

Three plots (FFS2, FFS5, and FFS6, hereafter called the FFS plots) were randomly located within larger areas that were burned in September or October of 2001 in separate fires, while two other plots (LOTHAR and UPTHAR, hereafter called the Tharp plots) were haphazardly placed within a larger prescribed fire unit that was burned in a single fire in October of 1990. Air temperatures during ignition ranged from 10 to 18 °C, relative humidity ranged from 20 to 63%, and wind speed ranged from 0 to 7 km/h. Average fuel moistures for litter and duff were ~18%, for 100-h fuels ~12% and for 1000-h fuels ~32%. All plots were burned by a combination of strip headfires and backing fires. Flame lengths were generally less than 1 m, with occasional flame lengths up to 2 m. Total pre-fire fuel loads were relatively high (~200 Mg ha^{-1}), composed primarily of large (>7.6 cm diameter [1000-h]) woody fuels, and litter and duff. Fuel consumption was high, with fires reducing total fuel loads to ~13% of pre-fire levels. Percentage of crown volume scorched (PCVS) within the plots was highly variable (within all plots: minimum PCVS = 0%, maximum PCVS = 100%; plot level average PCVS = 36 to 82%). See Knapp et al. (2005) and Mutch and Parsons (1998) for details.

Prior to burning all trees ≥1.37 m in height in the plots were tagged, mapped, measured for diameter at 1.37 m height (diameter at breast height, DBH), and identified to species; we had a total of 2236 trees for analysis. All plots were censused annually for tree mortality. We remeasured live tree stem diameter at the 5th year post-fire, which was used to calculate post-fire annual diameter growth increment. We recorded new recruitment (trees reaching 1.37 m in height) annually in the FFS plots, and at the 5th year post-fire in the Tharp plots. For consistency in methods across all plots we included the recruits in the data at the 5th year post-fire. Recruitment of trees was low in all plots up to five years after the fires (and likely represent stems that escaped the fire), with a total of 10 recruits across all plots (eight of which were in the FFS2 plot and the remaining two in the LOTHAR plot).

The goals for white fir mixed-conifer forest structure restoration at Sequoia National Park stipulate stand density to range between 50 and 250 trees ha^{-1} for stems <80 cm DBH, and 10 and 75 trees ha^{-1} for stems ≥80 cm DBH by the 5th year following initial treatment with prescribed fire (Fire and Fuels Management Plan, Sequoia and Kings Canyon national parks, *Unpublished report*). Additionally, forest composition goals call for treated stands to contain 40–80% *Abies*, and 15–40% *Pinus*, with the remainder made up from other species (target conditions for species composition were mostly in place prior to burning in our study plots, Table 1). The plan does not explicitly consider desired biomass retention amounts, nor does it consider the alteration of stem spatial arrangement as a management objective.

2.2. Data analysis

We assessed changes in forest structure in terms of stand density (stems ha^{-1}, including recruitment) and aboveground living stem biomass (Mg ha^{-1}, including post-fire growth), estimated from standard allometric equations tailored to the Sierra Nevada (Means et al., 1994). Biomass changes from growth were interpolated between stem diameter measurement years. We described trends in post-fire stand density and live stem biomass using mixed models (Pinheiro and Bates, 2004). Visual inspection of trends suggested that stand density and live stem biomass declined nonlinearly following the fires. We tested several model forms (e.g., linear, exponential decline, saturating) with and without random differences among plots, and with and without temporally correlated error terms. Selection using *AIC* (Burnham and Anderson, 2002) showed most support for modeling the trend in stand density as a mixed model with an asymptotic (Michaelis–Menten) regression function including temporally autocorrelated errors:

$$n_{ij} = \frac{(\beta_1 + b_{1i})\ \text{year}_j}{(\beta_2 + b_{2i}) + \text{year}_j} + \varepsilon_{ij}, \quad \text{where } b_{1i} \sim N(0, \sigma_1^2),$$

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

where n_{ij} represents the density of stems over the census interval and year_j represents the year of the jth census. Parameter β_1 represents the mean horizontal asymptote (where losses and gains in stem density are expected to balance) and parameter β_2 is the mean rate constant (which indicates how fast the asymptote is approached). The random effects among plots, b_{1i} and b_{2i}, are assumed to be normally distributed. Error was modeled as normally distributed independent noise (a_{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. Comparisons using *AIC* indicated the trend in aboveground live stem biomass (*bst*) was best modeled using an exponential decay regression function with random differences among plots and temporally autocorrelated errors:

$$bst_{ij} = (\beta_1 + b_{1i}) + \exp((\beta_2 + b_{2i}) \times \text{year}_j) + \varepsilon_{ij},$$

$$\text{where} \quad b_{1i} \sim N(0, \sigma_1^2), \quad b_{2i} \sim N(0, \sigma_2^2),$$

$$\text{and} \quad \varepsilon_{ij} = \phi\varepsilon_{i,j-1} + a_{ij} \tag{2}$$

where parameters β_1 and β_2 are fixed effects, and the plot effect (b_{1i} and b_{2i}) was assumed to follow a normal distribution. All parameters were estimated using maximum likelihood.

We tested changes in stem size distributions of each plot using a departure index, *M*, that is similar to the Gini coefficient, but distinguishes both the magnitude and direction of change (Menning et al., 2007). The departure index *M* for stem diameter size class distributions was calculated for pre-fire versus the 8th year post-fire censuses, and compared against a null model that assumed no differences between pre- and post-fire stem size class distributions. Differences in *M* between the observed values and null model were compared using a Wilcoxon signed rank test.

We calculated changes in the annual stand mortality rates, creating 95% confidence intervals from 1000 bootstrapped samples for the average mortality rate among plots for each year post-fire. The average annual mortality rate in the burned plots were compared against the annualized mortality rate from the two most recent complete censuses in seven unburned old-growth mixed conifer forest plots found in or near the Giant Forest (unburned plots names: LMCC, LOGSEGI, LOLOG, SUABCO, SUPILA, SURIP, UPLOG; see van Mantgem and Stephenson, 2007; van Mantgem et al., 2009). Using the two most recent censuses to determine unburned plot mortality rate creates a conservative comparison against the burned plots, as background mortality rates in the unburned plots have been increasing over time (van Mantgem and Stephenson, 2007; van Mantgem et al., 2009). Permutation tests were used to determine differences between the average annual mortality rate for each year post-fire and the annual morality rates in the unburned plots.

We described tree spatial distributions within plots by considering individual trees as points, and examined nearest neighbor patterns at multiple scales using the inhomogeneous *K*-function, a second-moment measure that accounts for variability in the average density (also called intensity) of trees across each plot (Baddeley, 2008). For each plot we considered spatial patterns within a search radius of up to 25% of the shortest side in any plot, typically 25 m (though for rectangular plots this distance was less). We determined the significance of the departure from complete spatial randomness (CSR) by comparing observed values to the 95% simultaneous critical envelope (Baddeley, 2008) [avoiding inflation of type I errors; Loosmore and Ford, 2006], calculated from 999 simulations of random point fields, with intensity patterns estimated using a kernel smoothing function. Results are presented in terms of the normalized *K* statistic, $L(t)$ (where t represents the search radius), which is centered on zero. $L(t)$ values within the 95% critical envelope are consistent with a random distribution of stems, while values below the critical envelope are considered significantly uniform, and values above the critical envelope are considered significantly clumped.

Simple comparisons of pre- and post-fire spatial patterns may be misleading. The estimated intensity patterns were different in the pre- and post-fire intervals, which potentially affects the estimates of spatial patterns. Moreover, if fire substantially reduces the number of trees, it will result in higher uncertainty and wider critical envelopes relative to the pre-fire interval. Consequently, detecting statistically significant departures from random spatial patterns may simply be more difficult in low density post-fire stands, even if fire has not altered the underlying spatial pattern of stems. To directly test the effect of fire we used a marked point process model (Baddeley, 2008) to compare the pattern of survivors eight years post-fire ($K(t)_{\text{post-fire}}$) to the pattern of trees prior to fire ($K(t)_{\text{pre-fire}}$). The expectation of the test statistic (i.e., $K(t)_{\text{post-fire}} - K(t)_{\text{pre-fire}}$) under random mortality is 0, with significant values greater than 0 indicating increased post-fire clumping and significant values less than 0 indicating increased post-fire uniformity. Significance was determined by values exceeding the 95% simultaneous critical envelopes created from 999 simulations using random labeling. Note that differences in *K*-functions are conditioned on the pre-fire spatial pattern, allowing us to use the homogeneous formulation of the *K* statistic. We used the same procedures to test changes in the spatial distribution of *A. concolor* ≤ 50 cm DBH. This stem diameter represents a cut point in Sierra Nevada mixed conifer forests where it is equally likely that individual stems established either before or after Euro-American settlement (Nathan Stephenson, *Unpublished data*). All analyses were conducted with R (R Development Core Team, 2009).

3. Results

By eight years post-fire prescribed fires reduced the average stem density by 67%, from 420 to 139 stems ha^{-1} (Fig. 1a). Average density by the eighth year post-fire for trees <80 cm DBH were reduced by 72% (from 370 to 104 stems ha^{-1}), while densities for stems ≥80 cm DBH were reduced by 30% (from 50 to 35 stems ha^{-1}), comparing favorably to local restoration targets. Prescribed fire resulted in only a 32% reduction in average aboveground live biomass (from 578 to 391 Mg ha^{-1}) by the eighth year post-fire (Fig. 1b), reflecting the retention of most large trees.

The preferential survival of large trees was also discerned by changes in stem size distributions, where post-fire distributions were shifted to the right (average $M = +0.301$, average range of $M = -0.299$ to 1.701). The right shift in stem size distributions was seen in all five plots, a change that was significantly different compared against a null model of no change (Wilcoxon signed rank test, $P = 0.0075$).

Prescribed fire did not lead to changes in the proportions of major species in terms of numbers of trees (average pre-fire proportions: *A. concolor* = 72%, *P. lambertiana* = 14%, other species = 14%; average eighth year post-fire proportions: *A. concolor* = 72%, *P. lambertiana* = 12%, other species = 16%; χ^2 test, $P = 0.83$). However, there was a suggestion that the representation of *A. concolor* declined in terms of live stem biomass (average pre-fire biomass proportions: *A. concolor* = 78%, *P. lambertiana* = 13%, other species = 9%; average eighth year post-fire biomass proportions:

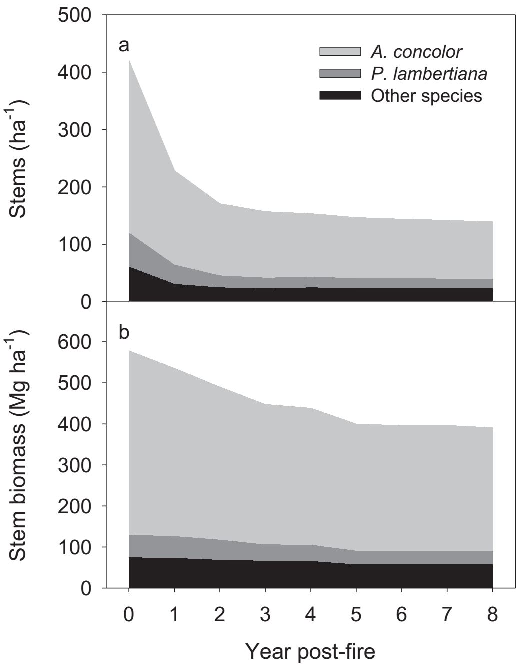

Fig. 1. Average annual changes to stand density (a), and live stem biomass (b) among five 1 ha monitoring plots in Sierra Nevada mixed conifer forests following prescription burning. Proportional changes to species composition were not significant in terms of stand density or biomass at $\alpha = 0.05$.

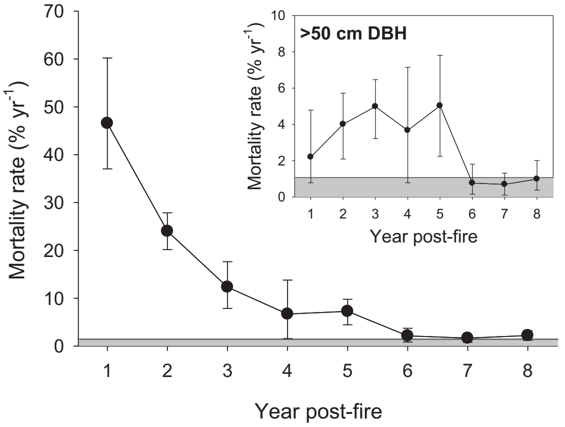

Fig. 2. Post-fire average tree mortality rate trends for five mixed-conifer forest plots. Error bars represent 95% confidence intervals calculated from 1000 bootstrapped samples. The shaded area corresponds to the average tree mortality rate in six nearby unburned stands (see text). The insert shows the same information for trees >50 cm DBH.

A. concolor = 71%, *P. lambertiana* = 17%, other species = 12%; χ^2 test, $P = 0.07$).

Stand density declined rapidly following burning while the reduction in stand biomass was gradual. Mixed model analysis showed a steep decline in stand density after the fires, with an expected average asymptote of stand density of approximately 138 trees ha^{-1} (Michaelis–Menten asymptote parameter, $\beta_1 = 137.8$, S.E. = 22.4, $P < 0.0001$), very close to the observed average stand density eight years post-fire (139 stems ha^{-1}). Our model of live stand biomass suggests more gradual changes in rates of biomass loss following fire (exponential decay slope parameter, $\beta_2 = -0.057$, S.E. = 0.009, $P < 0.0001$), with an expected average stand live stem biomass of 381 Mg ha^{-1} by the eighth year post-fire (close to the observed eighth year post-fire average live stem biomass, 391 Mg ha^{-1}).

The average annual mortality rate was 47% one year after the fires, but dropped to 2% by the eighth year after fire (Fig. 2). Mortality rates were significantly higher than nearby unburned forest plots (average unburned plot mortality rates = 1.5%) from one to six years post-fire (permutation tests, $P \leq 0.045$), after which mortality rates between burned and unburned plots were indistinguishable (permutation tests, $P \geq 0.27$). Overall mortality rates for the eight year post-fire interval were similar for *A. concolor* and *P. lambertiana* (13% year^{-1} and 15% year^{-1}, respectively). Trees that had likely established before Euro-American settlement (stems >50 cm DBH) showed a different response to fire, where mean mortality rates increased up to five years post-fire (although with high variability), after which mortality rates quickly returned to background levels (Fig. 2).

Prior to burning the stem spatial distributions in four of five plots showed significant clumping at small scales (1 to approximately 8 m), while all plots showed random distributions at intermediate scales (approximately 8–15 m), and uniform distributions at larger scales up to 25 m (Fig. 3a). Direct comparisons of the pre- and post-fire *K*-functions suggest that fire-caused mortality was spatially random, and prescribed fire did not fundamentally reconfigure spatial patterns (Fig. 3b). Two plots (LOTHAR and FFS6) did show increased uniformity at larger scales, but this change simply emphasized existing patterns. Prescribed fire did, however, reduce the occurrence of evaluations of statistical significance for small scale clumping (with only a single plot, FFS2, retaining statistically significant small scale clumping post-fire), an effect of increased widths of critical envelopes from smaller post-fire sample sizes (results not shown), not a fundamental change in the post-fire spatial pattern at small scales (Fig. 3b). Similar spatial patterns both before and after burning were found for *A. concolor* ≤50 cm DBH, the subset of trees most likely to have recruited following fire exclusion.

4. Discussion

Our results are consistent with the idea that prescribed fire can bring about structural changes that are consistent with restoration

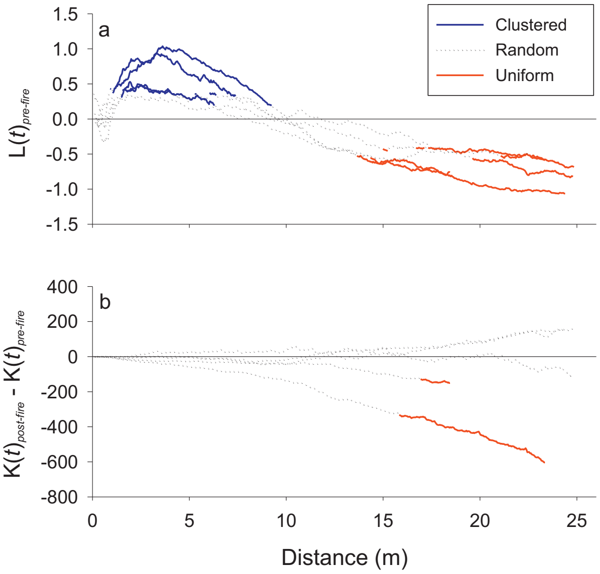

Fig. 3. Tree spatial pattern for the five mixed-conifer monitoring plots before burning (a), and the difference between pre- and post-fire spatial patterns (b). Lines are the observed normalized *K*-function [$L(t)$] or the differences in the *K*-function for post- and pre-fire intervals [$K(t)_{\text{post-fire}} - K(t)_{\text{pre-fire}}$] for live trees. Dashed intervals represent random distributions and solid intervals represent non-random distributions. Significance was determined by departures from 95% simultaneous critical envelopes (Baddeley, 2008).

goals in old-growth mixed conifer forests in the Sierra Nevada. With the exception of species composition, the broad restoration targets set by managers at Sequoia National Park were met using prescription fire. Small tree density (ladder fuels) and surface fuels were reduced (also see Mutch and Parsons, 1998; Knapp et al., 2005), offering a degree of protection from future catastrophic fires for the remaining stems (i.e., live carbon stocks) (van Wagtendonk, 1996; Stephens and Moghaddas, 2005; Hurteau et al., 2008). Although initial ("first-entry") prescribed fire treatments may not fully recreate presumed historic stand structure (North et al., 2007; Schwilk et al., 2009), it is possible that repeated application of prescribed fire could create more open forest conditions. The effectiveness of prescribed fire at driving changes in forest structure in our study might also be due to the use of relatively high severity fires (as measured by high fuel consumption) (Fulé et al., 2004), as well as the inclusion of long-term delayed tree mortality.

We would have underestimated fire effects had our observations stopped after one or two years post-fire. Responses in forest density were nonlinear over time, taking up to eight years to approach a presumed asymptote. We found even more gradual changes in the rate of live stem biomass loss, associated with elevated tree mortality rates (particularly for trees >50 cm DBH) up to six years post-fire. Delayed tree mortality, usually found in larger trees subjected to disturbance amplifiers such as bark beetles, is commonly observed when monitoring takes place for five years or longer (Hood, 2010) [although some long-term post-fire monitoring efforts fail to find substantial delayed mortality, e.g., Youngblood et al. (2009)]. Our findings support the idea that up to five years of post-fire monitoring is needed to avoid potentially underestimating the effects of delayed tree mortality. Observations past the fifth year post-fire did not reveal large changes.

Because large trees contain the vast majority of live tree carbon stocks, accounting for the delayed mortality of large trees will influence assessments of fire effects on forest carbon. Live stem biomass declined throughout post-fire observations, although the rate of biomass loss slowed considerably by eight years post-fire. While high severity disturbances can change forests from net carbon accumulators to net carbon emitters for long periods (Cohen et al., 1996; Kashian et al., 2006; Dore et al., 2010), our results imply that low to moderate severity disturbances have milder effects on live biomass, likely stabilizing in a relatively short period of time (likely less than 10 years) (also see Hurteau and North, 2010; Webster and Halpern, 2010). Moreover, reductions in forest density following prescribed fire are expected to decrease risks of crown fires (Stephens et al., 2009), potentially reducing the likelihood of massive carbon losses from future wildfires (Hurteau and North, 2009; North et al., 2009).

Prior to prescribed burning the spatial arrangement of stems in our sampled stands was heterogeneous in spite of more than a century of fire exclusion. Stem spatial distributions could be described as clumped, random, or uniform depending on scale. For the pre-fire interval our finding of significant clumping at small spatial scales is consistent with past results (Taylor, 2004; North et al., 2007), while our finding of random distributions at medium scales and uniform distributions at larger scales may be a result of using a more realistic test statistic (one that relaxes the assumption of homogeneity of stem densities within plots) coupled with more appropriate estimates of the critical envelopes [one that avoids inflating type I errors; Loosmore and Ford, 2006].

Fire-caused mortality appeared to be spatially random, and therefore cannot be responsible for major changes in forest pattern. Reductions in evaluations of statistically significant clumping at small scales following fire was a result of smaller sample sizes and wider critical envelopes, not a fundamental change in spatial pattern. Spatially random mortality was also observed for *A. concolor* $\leq$50 cm DBH, individuals that likely established after fire exclusion in the late 1800 s, which was unsurprising because a majority of individuals were in this class of trees. Spatially random fire-caused tree mortality may be a result of patchy within plot distributions of fire intensity, caused in turn by small scale variability in fuel distribution, topography and fire weather (Kilgore, 1973; Knapp and Keeley, 2006; van Mantgem and Schwilk, 2009). The creation and maintenance of spatial pattern represents the integration of the complex phenomena of individual mortality and recruitment events (Das et al., 2008), so the effects of prescribed fire on forest pattern cannot be characterized by examining only one or even a few burned plots. More study is needed to determine if prescribed fire mortality is spatially random in general, which could be accomplished in instances where large ($\geq$1 ha) study plots are used. Whether the restoration of spatial pattern should be a priority for prescribed fire treatments is debatable, given our imprecise knowledge of historic forest conditions (Stephenson, 1999) and the possibility that fire-caused mortality may be spatially random.

The complexity and variability of forest response to prescribed fire argues for more detailed and systematic investigations. This research effort will require understanding the mechanisms that drive differences in forest response to fire, and will need to consider the effects of climate, typical fire weather, alterations of fuel types and amounts, and physiological and allometric differences among dominant species. It will also become increasingly important to describe differences between first versus second-entry burns as restoration treatments continue at high priority sites (see Webster and Halpern, 2010). A mechanistic understanding of fire effects is critical, especially as climatic conditions change in ways that may preclude comparatively simple restoration prescriptions (Millar et al., 2007). These information needs underscore the commitment needed not only to execute the restoration treatments, but also for the continued monitoring of their effects.

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