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A comparison of effects from prescribed fires and wildfires managed for resource objectives in Sequoia and Kings Canyon National Parks

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ABSTRACT

Current goals for prescription burning are focused on measures of fuel consumption and changes in forest density. These benchmarks, however, do not address the extent to which prescription burning meets perceived ecosystem needs of heterogeneity in burning, both for overstory trees and understory herbs and shrubs. There are still questions about how closely prescribed fires mimic these patterns compared to natural wildfires. This study compared burn patterns of prescribed fires and managed unplanned wildfires to understand how the differing burning regimes affect ecosystem properties. Measures of forest structure and fire severity were sampled in three recent prescribed fires and three wildfires managed for resource objectives in Sequoia and Kings Canyon National Parks. Fine scale patterns of fire severity and heterogeneity were compared between fire types using ground-based measures of fire effects on fuels and overstory and understory vegetation. Prescribed fires and wildfires managed for resource objectives displayed similar patterns of overstory and understory fire severity, heterogeneity, and seedling and sapling survival. Variation among plots within the same fire was always greater than between fire types. Prescribed fires can provide burned landscapes that approximate natural fires in many ways. It is recognized that constraints placed on when wildfires managed for resource objectives are allowed to burn freely may bias the range of conditions that might have been experienced under more natural conditions. Therefore they may not exactly mimic natural wildfires. Overall, the similarity in fire effects that we observed between prescribed fires and managed wildfires indicate that despite the restrictions that are often placed on prescribed fires, they appear to be creating post-fire conditions that approximate natural fires when assessed on a fine spatial scale.

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1. Introduction

A century of fire exclusion has altered structure and processes in many forest types in the western U.S. (Caprio and Swetnam, 1995; Skinner and Chang, 1996; Fulé et al., 1997; Swetnam et al., 1999). Prescription burning has been widely utilized by fire managers as a management option for correcting the perturbations caused by previous fire suppression policy (Haines et al., 1998; Fernandez and Botelho, 2003). On contemporary landscapes human safety and control of prescribed burns are the highest priorities. Consequently prescribed burns are constrained by factors that may not simulate natural burning patterns, and could affect the extent to which this management practice can return forested ecosystems to naturally functioning systems. For example, prescription burning occurs under a narrower range of weather conditions than natural wildfires (U.S. Department of Interior, National Park Service, 2003; van Wagtendonk and Lutz, 2007). In addition, ignition patterns of prescribed burns may not mimic natural lightning ignitions in terms of the range of topographic patterns of ignition, season of burning or post ignition spread patterns and rates. To accomplish goals of fuel reduction and maintain control of fires, prescriptions often call for spot burning of unburned fuels, which, coupled with strip burning, may lead to more homogeneous burning patterns than would occur naturally. Due to air quality restrictions most prescribed burns have a very short ignition window during which smoke dispersal is forecast to occur. Thus these fires have prescriptions that ensure rapid progression and short burning times. This is quite unlike the much longer duration of naturally occurring wildfires (van Wagtendonk and Lutz, 2007).

In mixed conifer forests of the Sierra Nevada these constraints on prescribed fires could alter the ability of managers to reach forest restoration goals. Natural fire regimes in the Sierra Nevada are generally thought to be of mixed severity with most areas within a fire burning at low and moderate severity, though scattered patches of

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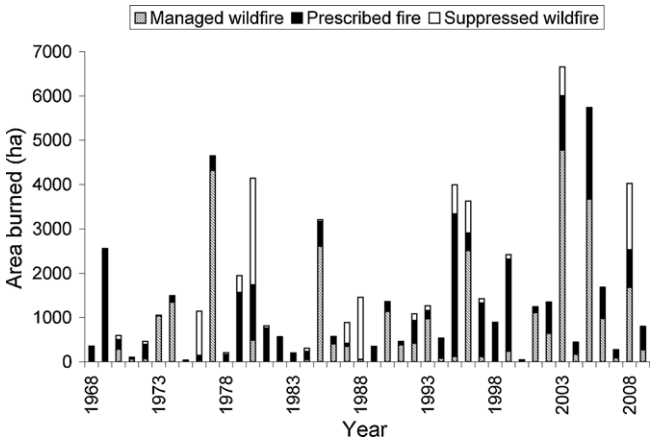

Fig. 1. Area burned by prescribed fires and wildland fires, both managed and unmanaged, since 1968 in Sequoia and Kings Canyon National Parks, CA (data from Sequoia–Kings Canyon Fire History Database).

high severity fire occur more frequently than previously thought (Stephenson et al., 1991; Beaty and Taylor, 2001, 2008; Collins and Stephens, 2010). These patterns produced heterogeneous landscapes that were critical to forest regeneration as well as a range of habitats for wildlife. Localized gaps in the forest canopy played an important role in providing suitable sites for tree regeneration when embedded in a landscape of surface fire that ensured survival of parent tree seed sources (Keeley and Stephenson, 2000). Likewise unburned patches were potentially critical for the recruitment of seedlings into older cohorts as well as providing seed sources for much of the understory biodiversity (Borchert et al., 2003; Bonnet et al., 2005; Lentile et al., 2007). Sustainable forest management may need to recreate such conditions in order to ensure processes such as tree recruitment and maintenance of natural biodiversity, of both plants and animals.

Since the late 1960s Sequoia and Kings Canyon National Parks have had a formal program of restoring fire to mid- and high-elevation forest types. This has involved managing natural lightning ignited fires and application of prescription burning. Over the past 35 years there have been nearly 600 prescribed burns that covered more than 20,000 ha within these parks (Fig. 1). In addition, since 1968 the National Park Service has allowed lightning fires to burn in portions of Sequoia and Kings Canyon National Parks in order to restore or maintain ecological conditions and processes (Kilgore and Briggs, 1972). Originally known as the "let-burn" program it has been renamed several times to "prescribed natural fires", "natural fires", "wildland fire use fires", and currently "wildfire managed for resource objectives" (Bancroft et al., 1985; U.S. Department of Interior, National Park Service, 2003). In this paper we call wildfire managed for resource objectives "managed wildfire". Since the beginning of the program, the parks have had 486 managed wildfires burn a total of 17,183 ha. Most of these fires have been relatively small (<4 ha), but there have been several much larger (>4000 ha) fires in both mid-elevation mixed conifer and higher red-fir (*Abies magnifica* A. Murray) forests in the parks. The unique management history in Sequoia–Kings Canyon National Parks makes it possible to match prescribed burns and managed wildfires by forest type, elevation, aspect and age, making these parks one of the few places one could do extensive comparative studies of how managed wildfires differ from prescribed fires. Comparison studies looking at spatial burn patterns and fire severity impacts of prescribed fires and managed wildfires could provide us with important benchmarks for understanding how the differing burning regimes affect ecosystem properties.

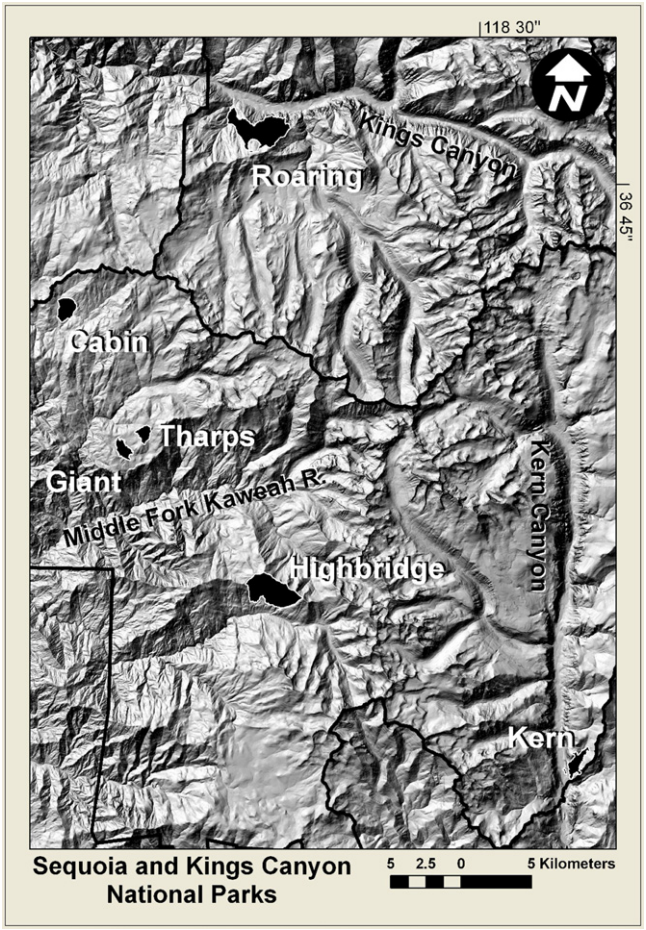

Fig. 2. Location of the fires that were sampled in Sequoia and Kings Canyon National Parks, CA. Prescribed fires were Cabin, Tharps, and Highbridge. These were each paired with a similar managed wildfire: Roaring, Giant, and Kern, respectively.

The objective of this study was to compare fine scale patterns of forest structure, burn severity, and patterns of burn heterogeneity between prescribed fires and managed wildfires in Sequoia and Kings Canyon National Parks. Our expectations were: (1) Managed wildfires would result in a more diverse understory stand structure evidenced by greater surviving shrub cover and greater abundance of large woody fuels than prescribed fires. (2) Managed wildfires would result in a greater proportion of unburned area within the fire perimeter. (3) Managed wildfires would result in a greater proportion of high severity burning than prescribed fires. (4) The variance in fire severity would be greater in managed wildfires than in prescribed fires. (5) Seedling and sapling survival would be greater in managed wildfires than in prescribed fires due to an increase in unburned patches.

2. Methods

2.1. Study area and site selection criteria

The study sites were located in Sequoia and Kings Canyon National Parks on the west side of the southern Sierra Nevada (Fig. 2). This area is characterized by a Mediterranean climate of wet winters and dry summers. Mean annual precipitation at 2000 m elevation averages 1400 mm (Stephenson, 1988; van Mantgem et al., 2006). Most of this is deposited in the form of snow. We limited our site selection to fires that were >40 ha and had occurred in the mixed conifer zone, including areas of giant sequoia-mixed conifer forest. Species composition was dominated by white fir (*Abies con-*

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color [Gord. & Glend.] Lindl. Ex Hildebr.). Other common species included Jeffrey pine (*Pinus jeffreyi* Balf.), red fir, and sugar pine (*Pinus lambertiana* Douglas). No other species accounted for >5% of the measured trees within any site.

Spatial patterns of fire severity are influenced by many factors including fire history, vegetation type, rate of fire spread, topography, and weather conditions (Turner et al., 1994; Collins et al., 2006; Beaty and Taylor, 2008; Haire and McGarigal, 2009). To minimize the effects of fire history on the results only fires that occurred in areas that had not experienced a previous fire over the past 80 years were used. In addition, only fires that had occurred within the last ten years were considered to ensure that fire effects would still be apparent. GIS layers of elevation, aspect, vegetation type, and year of ignition were used to match similar prescribed fires and managed wildfire pairings for comparison. The criteria used to generate potential fire pairings were: each pair must have been within 500 m elevation of each other, the dominant forest type, based on the USGS-NPS vegetation type classification map (http://biology.usgs.gov/npsveg/seki/index.html), must have been the same, and each fire must have burned during the same year.

The three selected prescribed fires were Cabin, Tharps, and Highbridge. They were each paired with a similar managed wildfire: Roaring, Giant, and Kern, respectively. Cabin prescribed fire burned 176 ha from 8 to 10 November 2006. Ignition was accomplished using drip torches and areal ignitions and resulted in a combination of backing and head fires. It was paired with the Roaring managed wildfire which was ignited by lightning on 23 July 2006 and burned 665 ha. Both burn areas were dominated by white fir and ranged in elevation from 2077 m to 2325 m and 2146 to 2495 m, respectively. Tharps prescribed fire burned 92 ha from 11 to 13 August 2003. Ignitions were accomplished using drip torches. The Giant managed wildfire was started by lightning on 30 July 2003 and burned 111 ha. Both fires were located in giant sequoia (*Sequoiadendron giganteum* [Lindl.] J. Buchholz) mixed conifer forest where white fir was the most abundant tree species within the plots. Elevation of the fires ranged from 2241 m to 2486 m and 1996 to 2143 m, respectively. After data collection had been completed, it was determined that 2 of the 25 subplots within the Giant fire had burned in 1981, but they were quite similar to the other subplots within the fire and their inclusion in the analysis did not alter any of the results. Highbridge prescribed fire burned 614 ha from 6 to 15 October 2006. Ignitions were accomplished using drip torches and areal ignitions. Kern managed wildfire began on 27 October 2005 from lightning and burned 140 ha. Both fires were located in Jeffrey pine – white fir woodland, though white fir was the most abundant species in the plots sampled in Highbridge (75%) while Jeffrey pine was most abundant at Kern (55%). Elevations ranged from 2269 m to 2708 m and 2157 to 2419 m, respectively.

2.2. Sampling design and data collection

Data collection occurred from June through August 2008 and focused on sampling various measures of forest structure, fire severity and heterogeneity. When the burn size differed between the paired prescribed fire and managed wildfire, the area of the smaller of the two fires was used to restrict sample area within the larger fire to equalize the size of the area sampled within them. Fire size has been linked to differences in fire severity and heterogeneity (Turner et al., 1994; Haire and McGarigal, 2009). These differences may be due to a host of factors including rate of fire spread, more diverse topography in larger areas, or an increased number of vegetation types within larger fires so that larger fires tend to display a wider range of burn severities than smaller fires (Turner et al., 1994; Lutz et al., 2009). We restricted the size of our sampling area within the larger fire so that any variability in the data was not due purely to sampling a larger area within a particular fire type. In addition, many of the managed wildfires were in relatively remote locations and randomly sampling across the entire burn area was logistically difficult. By randomly sampling in a restricted area we reduced the time potentially required to sample remote plots. The area that was selected for sampling within the larger fire was based on Landsat imagery to roughly mimic the range of large scale burn severity within the paired fire area. Five 100 m × 100 m plots were randomly located within each burn. Within a burn, only forested areas were included in the potential sample area, excluding lakes, large areas of rock or bare ground, and shrub fields. Within these plots, five 0.1 ha circular subplots were established at the four corners and center of each plot.

Within each subplot, the center point location was recorded using a handheld GPS and slope and aspect were measured. Several measures of forest structure including live tree species composition and density, number of snags, surviving seedling and sapling density, large woody fuels density, and live and dead shrub cover, were recorded. Each tree was identified by species, and dbh, height, base of live crown height, and health status (live, dead, dead before fire) were recorded. Data were collected on 3545 trees (2793 live and 752 dead) across the six fires. Tree seedlings (<1.37 m tall) and saplings (dbh < 10 cm) that survived the fire were tallied by species within each subplot. Regeneration that occurred since the fire was not included in these counts. When needed, growth whorls were used to determine whether seedlings had germinated pre- or post-fire. The total percent cover of understory species was visually estimated. Large down logs (>7.6 cm diameter) that were not consumed by the fire were tallied within each subplot.

Several measures of fire severity including crown scorch, stem char, percentage ground surface area burned, and tree and shrub mortality were also recorded. Crown scorch and stem char were recorded for all trees taller than 1.37 m. To assess crown scorch severity, we visually estimated crown volume scorched and measured maximum scorch height using an MDL LaserAce® hypsometer. In older fires, when most dead needles had already dropped from the trees, we used dead branches as our estimate of pre-fire live crown in order to estimate percent crown volume scorched. Bole char was still readily apparent several years following fire. Bole char was assessed by measuring maximum char height on the stem using the hypsometer. The percentage ground surface area that was burned in the fire was visually estimated in each subplot and then adjusted to account for the "unburnable" area such as rocks or streams. It therefore represented the proportion of burnable area burned within a subplot.

Observations of crown scorch, stem char, litter and duff consumption, and evidence of burned understory vegetation such as charred stems, scorched foliage, or holes from burned out roots were used to assign a burn severity rating to each subplot. The rating was calculated by first dividing a subplot into eighths and rating each octant on a scale from zero to three (0 = unburned, 1 = low severity, 2 = moderate severity, and 3 = high severity). Fire severity categories were defined based on criteria similar to those used by the Fire Effects Monitoring and Inventory System (Lutes et al., 2006). An octant was considered unburned if there was no evidence of fire within the octant. This rating would have been applied to octants that burned, but where the effects were so minor that they were no longer apparent at the time of post-fire sampling as well. Low severity was defined as an octant where some pronounced fire effects such as stem char or litter and duff consumption were apparent, but the changes to the overstory in terms of crown scorch or mortality were small (<35% crown scorch on average). Moderate severity was defined as an octant where fire effects were prevalent throughout the area (>75% of the ground appeared to have burned based on consumption of duff litter, and understory trees and shrubs). In addition, individual tree mortality was common, but was limited to <50% of the overstory trees within the octant.

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Table 1
Stand structure of six prescribed and managed wildfires in Sequoia and Kings Canyon National Parks, CA. This table shows average values within 25 0.1 ha subplots within each fire. Live trees, dead trees, and large woody fuel are counts within a subplot.

Fire	Cabin prescribed fire[a]	Roaring managed wildfire[a]	Tharps prescribed fire[a]	Giant managed wildfire[a]	Highbridge prescribed fire[a]	Kern managed wildfire[a]
dbh (cm)	37.1 (6.7)	39.5 (7.0)	51.0 (12.1)	61.5 (20.2)	50.1 (10.4)	46.8 (10.5)
Height (m)	15.9 (3.4)	17.1 (1.5)	20.9 (6.3)	23.7 (9.0)	17.0 (3.3)	17.4 (4.1)
Live trees	31.6 (17.2)	22.4 (8.4)	15.1 (2.0)	11.2 (6.0)	16.6 (11.0)	15.0 (6.6)
Dead trees	6.6 (7.8)	8.2 (7.8)	3.9 (2.3)	5.6 (4.5)	3.8 (3.6)	2.2 (1.1)
Shrub cover (%)	6.4 (6.9)	5.8 (7.1)	6.7 (2.2)	9.5 (10.9)	5.1 (2.7)	4.7 (3.8)
Large woody fuel	24.4 (11.3)	28.0 (15.8)	16.7 (1.2)	22.0 (9.6)	14.0 (9.2)	12.1 (5.6)
Surviving regeneration	271.0 (254.7)	377.8 (427.5)	91.0 (45.3)	133.4 (140.6)	85.4 (81.9)	69.6 (66.1)

ABCO: *Abies concolor*, PIJE: *Pinus jeffreyi*. Values in parentheses are standard deviations.
[a] Fire type.

High severity was defined as an octant where >50% of the overstory trees had been killed by the fire. Little to no understory vegetation remained and nearly all litter and duff had been consumed. These eight ratings were then averaged together to produce the average subplot visual fire severity rating to capture fine scale differences in fire severity within a plot.

2.3. Analysis

Analysis was conducted using the nlme package (Pinheiro et al., 2009) and lme4 package (Bates and Maechler, 2009) within the R statistical package version 2.8.1 (http://www.r-project.org/). We used generalized mixed effects (GLMM) regression models to account for the hierarchical structure of our field data. By treating plot as a random effect, we accounted for the potential correlation between subplots nested within the same plot (Zuur et al., 2009). In mixed effect models, observations between levels are independent, but observations within each level are dependent producing two sources of variation (Demidenko, 2004). In our GLMM models plot and fire were nested random effects assumed to follow a multivariate normal error distribution. Fire type (prescribed vs. managed wildfire) was treated as a fixed effect. We used the mean dbh, height, surviving shrub cover, as well as cumulative live and dead tree density, and abundance of large woody fuels (LWF) within subplots to compare post-fire stand structure between prescribed fires and managed wildfires. The log of dbh, height, surviving regeneration, and LWF were used in the analysis to meet assumptions of normality in the error distributions. These variables were analyzed using linear mixed effects regression models (LME). Each individual fire pair was also examined using LME models where plot was again treated as a random effect.

Multiple criteria including visual burn severity rating, percentage ground surface area burned, tree mortality rate, percent crown volume scorched, and maximum stem char height were used to assess whether fine scale patterns of burn severity differed between prescribed fires and managed wildfires. The log of maximum char height was used in the analysis to meet assumptions of normality. The effect of tree size on post-fire mortality and whether this differed between fire types was tested using GLMM. In this model, individual trees were used as the sample unit and subplot was nested within plot within fire and all were treated as random effects. Post fire mortality was a binary variable (dead, live) so we used a mixed effects logistic regression model following a binomial distribution. An interaction between fire type and dbh was included to test whether the effect of dbh on mortality differed by fire type. A change in Akaike Information Criterion (AIC) >2 was used to evaluate whether the removal of a fixed effect or interaction substantially reduced model fit (Burnham and Anderson, 2002). The effect of fire severity on mean surviving seedlings and saplings was tested using a mixed effects regression model where plot and fire were treated as nested random effects and visual fire severity rating and fire type were fixed effects. The inclusion of an interaction term between fire type and fire severity rating on model fit was evaluated using AIC.

To test for differences in fire severity heterogeneity, each measure of fire severity at the subplot level was used to calculate a plot level standard deviation and plot was treated as the sample unit. The mean of these standard deviations was then compared between fire types using mixed effects regression where fire was treated as a random effect. For the pairwise comparisons of fire heterogeneity, Wilcoxon rank sum tests were used to test for differences between fire types.

3. Results

Stand structure was similar between fire types and between individual fire pairings. Average tree dbh was comparable between prescribed fires and managed wildfires averaging 40.3 cm and 43.8 cm, respectively (LME, $P = 0.828$; Table 1). Dbh ranged from 10 cm to 182 cm in prescribed fires compared to 10–562 cm in managed wildfires and the proportion of small trees (<20 cm dbh) was greater in prescribed fires (34%) than managed wildfires (29%). The largest difference in tree dbh among pairings was between Tharps (51.0 cm) and Giant (61.5 cm), but this difference was not statistically significant (LME, $P = 0.435$). There was no detectable difference in average tree height between fire types (LME, $P = 0.610$; Table 1), though it tended to be slightly higher in managed wildfires (17.2 m) compared to prescribed fires (15.4 m), with the largest difference being between Tharps (20.9 m) and Giant (23.7 m). Average live tree density ranged between a low of 112 trees ha^{-1} in Giant and a high of 316 trees ha^{-1} in Cabin (Table 1). All of the prescribed fires had slightly higher live tree densities than the managed wildfires, averaging 211 and 162 trees ha^{-1}, respectively, but this difference was not statistically significant (LME, $P = 0.538$). Dead tree density was also quite similar between pairings and fire types as prescribed fires averaged 47 dead trees ha^{-1} compared to 53 dead trees ha^{-1} for managed wildfires (LME, $P = 0.988$; Table 1). Surviving shrub cover did not differ significantly between any of the fire pairings or between fire types, averaging 6.1% in prescribed fires compared to 6.7% in managed wildfires (LME, $P = 0.579$; Table 1). Abundance of large woody fuels was similar between prescribed fires and managed wildfires as well averaging 184 ha^{-1} and 207 ha^{-1} in prescribed fires and managed wildfires, respectively (LME, $P = 0.876$; Table 1). The mean number of surviving seedlings and saplings varied dramatically among subplots, even within the same plot with counts of surviving regeneration ranging between 0 and 1365. There was not a significant difference in the mean number of seedlings and saplings per subplot between fire types (LME, $P = 0.955$) or fire pairings (Table 1). The largest absolute difference occurred between Cabin and Roaring, which averaged 271.0 and 377.8 surviving seedlings and saplings per subplot, respectively. However, because of the large variance in density among subplots, this difference was not statistically significant (LME, $P = 0.774$).

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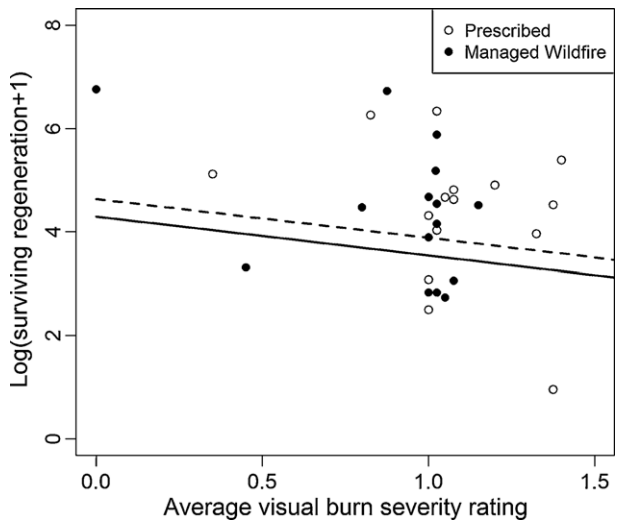

Fig. 3. The effect of fire severity on abundance of seedlings and saplings that survived prescribed or managed wildfire in Sequoia and Kings Canyon National Parks, CA using mixed effects linear regression. Fire type (prescribed vs. managed wildfire) and visual fire severity rating (0 = unburned, 1 = low severity, 2 = moderate severity, 3 = high severity) were fixed effects and plot and fire were treated as nested random effects in the model to account for the hierarchal structure of the data. The log of surviving seedling regeneration was used to meet assumptions of normality of the residuals.

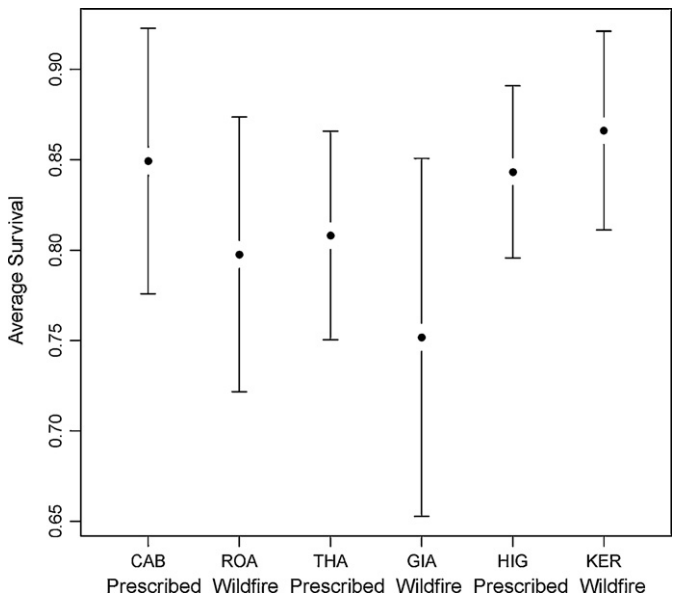

Fig. 4. Average proportion of trees that survived following prescribed or managed wildfire in Sequoia and Kings Canyon National Parks, CA fire. Lines are 95 percent confidence intervals. Pair 1 is Cabin Creek (CAB) and Roaring (ROA), pair 2 is Tharps (THA) and Giant (GIA), and pair 3 is Highbridge (HIG) and Kern (KER).

There was a negative trend in surviving seedling and sapling density per subplot with increasing visual burn severity rating (LME, $P = 0.076$). There was a mean of 413 seedlings and saplings per subplot in unburned subplots, 140 seedlings and saplings in subplots that burned at low severity, and 153 seedlings and saplings in subplots that burned at moderate severity (Fig. 3). The effect of visual fire severity rating on surviving regeneration density did not differ between fire types (ΔAIC = 0.5).

There was no significant difference in burn severity between prescribed fires and managed wildfires regardless of which estimate of fire severity was used (visual severity rating, percentage ground surface area burned, tree mortality rates, average crown volume scorched, and average maximum stem char height; Table 2). The visual severity rating of 103 of the 150 subplots sampled within the burns was unburned or low severity and there were no subplots that burned at high severity. There was no statistical difference in burn severity rating between prescribed and managed wildfire as the mean burn severity rating for prescribed fires was 1.07 compared to 0.90 for managed wildfires (LME, $P = 0.182$, Table 2). When analyzed separately, the Highbridge–Kern pairing did show a significant difference in fire severity rating with a mean rating or 1.24 at Highbridge (prescribed fire) compared to 0.98 at Kern (managed wildfire) (LME, fire type coefficient = −0.2557143, SE = 0.092, $P = 0.024$), though this result was no longer significant at the 95% confidence level when the familywise error rate was adjusted using Bonferroni's procedure to account for multiple comparisons.

The similarity in fire severity was also reflected in the estimated percentage of ground surface area burned (Table 2). The mean in the prescribed fires (55.7%) was lower than that of the managed wildfires (64.5%) but this difference was not significant (LME, $P = 0.408$). Mean tree survival was also similar between prescribed fires and managed wildfires (83% and 81%, respectively, LME, $P = 0.546$). Survival rates were not statistically different between any of the three fire pairings with the largest difference occurring between Tharps (80.8%) and Giant (75.5%) (Table 2). Mean tree survival often differed dramatically between subplots, even within the same fire with some subplots losing very few trees while others lost over half (Fig. 4). Percentage of the crown volume that was scorched by the fire was quite similar between prescribed fires and managed wildfires with crown volume scorched averaging 18.9% and 21.0%, respectively. This was also true between the individual fire pairings with the largest difference being between Cabin (15.8%) and Roaring (27.2%) (Table 2). There was no significant difference

Table 2
Measures of fine scale fire severity within six prescribed and managed wildfires in Sequoia and Kings Canyon National Parks, CA. Severity rating is the average subplot visual severity rating where 0 = unburned, 1 = low severity, 2 = moderate severity, and 3 = high severity. Ground burned is the average area within a subplot that was burned during the fire. Survival is the average percentage of trees that survived. Crown Vol. Scorch is the average crown volume scorched and max char height is the maximum height on the stem that was charred by the fire.

Fire	Fire type	Severity rating	Ground burned (%)	Survival (%)	Crown Vol. Scorch (%)	Max char height (m)
Cabin	Prescribed	0.92 (0.37)	39.8 (20.6)	84.9 (10.7)	15.8 (12.5)	1.0 (0.8)
Roaring	Managed wildfire	0.81 (0.46)	57.4 (37.0)	79.8 (15.3)	27.2 (19.5)	1.6 (1.1)
Tharps	Prescribed	1.07 (0.08)	57.3 (11.6)	80.8 (5.7)	18.6 (4.7)	1.2 (0.6)
Giant	Managed wildfire	0.91 (0.26)	65.4 (32.3)	75.5 (18.6)	18.5 (11.8)	2.4 (1.0)
Highbridge	Prescribed	1.24 (0.18)*	69.9 (13.1)	84.3 (6.6)	22.3 (11.4)	1.7 (0.9)
Kern	Managed wildfire	0.98 (0.11)*	70.6 (18.6)	86.6 (6.8)	17.3 (7.7)	2.2 (1.0)
All	Prescribed	1.07 (0.16)	55.7 (15.1)	83.4 (2.2)	18.9 (3.3)	1.3 (0.4)
All	Managed wildfire	0.90 (0.09)	64.5 (6.6)	80.6 (5.6)	21.0 (5.4)	2.1 (0.4)

Values in parentheses are standard deviations.
* A significant difference ($P < 0.05$) between fires.

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Table 3
Measures of fire severity heterogeneity within six prescribed and managed wildfires in Sequoia and Kings Canyon National Parks, CA. This table shows the average standard deviation (SD) of the plots within each fire. Severity rating is the average SD subplot visual severity rating where 0 = unburned, 1 = low severity, 2 = moderate severity, and 3 = high severity. Ground burned is the average SD of area within a subplot that was burned during the fire. Survival is the average SD of the percentage of trees that survived. Crown Vol. Scorch is the average SD of crown volume scorched and Max Char Height is the SD of maximum height on the stem that was charred by the fire.

Fire	Fire type	Severity rating	Ground burned (%)	Survival (%)	Crown Vol. Scorch (%)	Max char height
Cabin	Prescribed	0.48 (0.12)*	21.3 (9.0)	12.1 (6.8)	14.2 (8.3)	0.72 (0.39)
Roaring	Managed wildfire	0.12 (0.18)*	12.1 (8.9)	10.6 (6.9)	11.0 (8.9)	0.63 (0.54)
Tharps	Prescribed	0.09 (0.10)	18.7 (11.2)	11.7 (5.0)	15.1 (4.9)	0.72 (0.45)
Giant	Managed wildfire	0.22 (0.37)	19.0 (12.7)	15.0 (12.2)	16.0 (8.1)	1.88 (0.86)
Highbridge	Prescribed	0.19 (0.12)	15.1 (8.1)	7.7 (3.5)	12.9 (7.4)	0.78 (0.46)
Kern	Managed wildfire	0.14 (0.15)	18.7 (10.5)	10.9 (8.9)	13.9 (7.0)	1.40 (0.70)
All	Prescribed	0.26 (0.20)	18.4 (3.1)	10.4 (2.5)	14.1 (1.1)	0.74 (0.04)
All	Managed wildfire	0.16 (0.05)	16.7 (3.9)	12.2 (2.5)	13.6 (2.5)	1.30 (0.63)

Values in parentheses are standard deviations.
* A significant difference ($P < 0.05$) between fires within a pairing or between fire type.

in maximum stem char height between fire types as prescribed fires averaged 1.3 m compared to 2.1 m in managed wildfires (LME, $P = 0.363$).

Fire damage in the areas we measured was primarily limited to the understory trees and vegetation. Tree size was a significant factor controlling post-fire mortality as the proportion of small to medium size trees (<50 cm dbh) that were dead following the fire was 27% compared to 5% for large trees (≥50 cm dbh). There was a significant interaction between tree size and fire type, however, indicating that the effect of tree size on mortality from fire differed between fire types (coefficient of interaction term of dbh × fire type = 0.024, SE = 0.007, $P < 0.001$). The predicted probability of survival of a 10 cm tree in a prescribed fire was 71% compared to 53% for managed wildfire (Fig. 5). Probability of survival was almost identical between fire types for large trees as survival of a 50 cm tree was 93% for prescribed fires compared to 94% for managed wildfires.

Fire severity heterogeneity did not differ significantly between prescribed fires and managed wildfires for any of the measures of fire severity (Table 3). The only difference in fire severity heterogeneity was between the visual severity rating of Cabin (prescribed fire) and Roaring (managed wildfire) (Wilcoxon rank sum test, $W = 23$, $P = 0.036$, Table 3). This difference was not significant at the 95% confidence level after the familywise error rate was adjusted using Bonferroni's procedure to account for multiple comparisons.

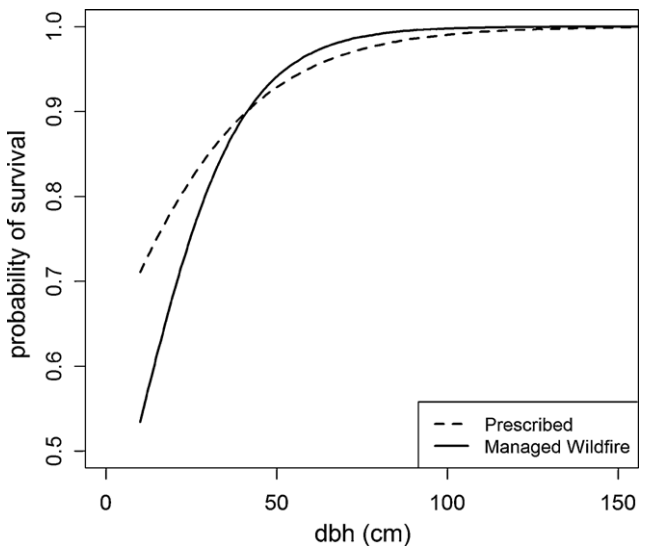

Fig. 5. Effect of tree size on probability of survival for prescribed and managed wildfires in Sequoia and Kings Canyon National Parks, CA using mixed effects logistic regression. Fixed effects were fire type, tree size, and their interaction. Random effects were subplot, nested within plot, nested within fire.

Mean plot standard deviation of percentage ground surface area burned, tree survival, percentage crown volume scorched, and max char height, were all similar between prescribed and managed wildfires (Table 3). In almost all of the measures of fire severity heterogeneity, the variance was greater among plots within the same fire than between fires of different fire types.

4. Discussion

The use of prescribed fires has proven to be an essential tool for forest restoration in the Sierra Nevada (Brown et al., 2004; Agee and Skinner, 2005). Yet, there are questions about whether prescribed fires are mimicking natural fire processes leading to successful restoration. In particular, prescribed fires may differ from natural fires in their fine scale heterogeneity and understory structure. Wildland fire management attempts to provide burned landscapes that approximate natural fires. It is recognized that constraints placed on managed wildfires may bias the range of conditions that might have been experienced under more natural conditions. For example fewer of these fires are allowed to burn under severe weather conditions or when smoke might impact local communities than might have happened naturally, thus potentially diminishing the extent of high severity burning. On the other hand these fires are now burning on landscapes that have missed one or more fire cycles and this would tend to produce more intense fires than occurred naturally, though this is likely not the case at higher elevations where fire return intervals are longer. Managed wildfires provide a reasonable approximation of natural fires that burn at moderate to low severity and can provide us with important benchmarks for evaluating the effectiveness of prescribed burns in mimicking natural fires. Patterns in stand structure and fire effects of natural wildfires and prescribed fires within each fire type have received much attention (Fulé et al., 2004; Collins and Stephens, 2007; Lentile et al., 2007; Keane et al., 2008); yet, rarely have these patterns been directly compared between fire types. By comparing fine scale patterns of forest structure, fire severity, and fire severity heterogeneity, we can gain a better understanding of how these different types of fires affect ecosystem properties. Managed wildfires and prescribed fires appeared to produce similar fine scale patterns of fire severity and heterogeneity in Sequoia and Kings Canyon National Parks.

Stand structure is a major determinant of ecosystem function as it influences net primary productivity, hydrology, erosion, wildlife use, and many other properties (Spies, 1997; Skinner, 2002; Knapp et al., 2005). One of the main restoration goals of prescribed fires is to create conditions that mimic natural fires by creating standing snags, altering the abundance of large woody fuels, and reducing live tree density, particularly smaller size classes (U.S. Department of Interior, National Park Service, 2003). Stand characteristics including species composition, density of live and dead

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trees, as well as tree size were all similar between prescribed fires and managed wildfires. Understory characteristics including shrub cover and amount of large woody debris was also similar between fire types. However, small tree survival was greater in prescribed fires than managed wildfires (Fig. 5).

Surviving seedling and sapling density was similar between prescribed fires and managed wildfires. The effect of fire severity on the density of surviving regeneration was also similar between fire types. While seedling and sapling mortality is often high following fire, those that survive can be an important component of forest regeneration and understory stand structure (Bailey and Covington, 2002). Most of the surviving seedlings and saplings occurred in unburned patches and because the unburned area was fairly similar between fire types, it is not surprising that the surviving regeneration density was similar as well.

One assumption about prescribed fires and managed wildfires is that because of the restrictions placed on the conditions under which a prescribed fire can occur, prescribed fires will result in lower severity than managed wildfires. This is because many prescribed fires are conducted during conditions that will limit fire intensity to safeguard against escape and promote low to moderate severity fire (DeBano et al., 1998). Wildfires on the other hand often ignite under conditions conducive to higher severity fire, often during times of extreme fire weather and low fuel moisture leading to more severe fires than prescribed fires (van Wagtendonk and Lutz, 2007). In this study, we found that fire severity was similar between fire types. This result was repeated over multiple measures of fire severity including crown volume scorch, stem char height, tree mortality rates, percentage ground surface burned, and visual fire severity rating. One possible explanation of why we did not find differences in fire severity between fire types is that prescribed fires and managed wildfires often differ dramatically in their total area burned as managed wildfires tend to be much larger than prescribed fires. This may lead to landscape level differences in severity between prescribed and managed wildfires (van Wagtendonk and Lutz, 2007) that we did not measure. Another potential reason is that our plots were not in areas of predominantly high severity fire within the fires that we sampled and while we found that stand structure and fire effects in low and moderate severity fire did not differ between prescribed and managed wildfires, this may not be the case for high severity areas. Both prescribed fires and managed wildfires resulted in unburned patches as well as gaps created by tree mortality from fire. These patches were scattered within a matrix that experienced predominantly low severity fire. The area of unburned patches as well as density of fire killed trees caused by higher intensity fire was similar between fire types. Our expectation that managed wildfires would contain both more unburned areas as well as areas that burned under high severity was not supported. We expected that prescribed fires might also result in greater percentage of area burned, but this was not the case.

Within stand fire severity and heterogeneity is driven by patterns of connectivity in surface fuels and stochastic variation of weather driven burning conditions (Collins et al., 2009). Modeling studies suggest fuel connectivity may be critical to determining total fuel consumption and fire size (Miller and Urban, 2000). Empirical studies show clearly that heterogeneity in burning and the extent of unburned patches is important for tree regeneration and long term maintenance of biodiversity (Halpin, 1995; Keeley and Stephenson, 2000; Rocca, 2004). Our expectation that fire severity would be more variable in managed wildfires than prescribed fires, resulting in greater fine scale heterogeneity in stand structure and burn pattern was not supported by the data. We did not find a significant difference in the variation of fire severity between fire types, regardless of which measure of fire severity was used. Fire severity varied more among plots within the same fire than it did between fires, which reflected the patchiness in fire severity that occurred within both prescribed fires and managed wildfires.

As with most studies that investigate the effects of wildfire, we were limited by the fact that we did not have pre-fire data on forest fuels, composition, or structure. This limits the type of assertions we can make. For example, while we found that post-fire forest structure was similar between prescribed and managed wildfires, it is uncertain how pre-fire conditions may have influenced these results. Though we tried to control for multiple factors that influence forest structure and response to fire such as fire history, aspect, and vegetation type, we cannot be certain that the pre-fire conditions of our paired prescribed and managed wildfires were identical. We tried to account for this by developing selection criteria for the fire pairings that would result in pre-fire conditions being as similar between fire types as possible. However, this process limits our scope of inference because we only looked at recent fires in a relatively narrow elevation range to try and control for differences in pre-fire conditions. This issue is not unique to this study, however (van Mantgem and Schwilk, 2009).

4.1. Conclusions

This study allows us to begin to establish benchmarks for prescribed fires based on the fine scale patterns of understory structure, burn severity, and heterogeneity that occur in natural fires. Fuel loadings and other benchmarks have already been established and are widely used in setting prescribed fire objectives, but there has been little information on fine scale burn patterns (Keifer et al., 2000). This study allows us to begin to understand the amount of fine scale heterogeneity in burn severity we can expect in managed wildfires and how this affects ecosystem properties. In general, we found that prescribed fires and managed wildfires produced similar fine scale burning patterns and it appears that the prescribed fires in Sequoia–Kings Canyon National Parks are mimicking the burn patterns of managed wildfires quite well. This was due in part to the large variation in burn patterns within fires as the range of fire severity was greater within individual fires than between fire types. However, managed wildfires are only an approximation of natural wildfires because of the limited conditions under which they are allowed to burn. Overall, the similarity in fire effects that we observed between prescribed fires and managed wildfires indicate that despite the restrictions that are often placed on prescribed fires, they appear to be creating post-fire conditions that approximate natural fires when assessed on a fine spatial scale.

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