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Fuel treatment effectiveness in California yellow pine and mixed conifer forests

H.D. Safford [a,b,*], J.T. Stevens [c], K. Merriam [d], M.D. Meyer [e], A.M. Latimer [c]

[a] USDA Forest Service, Pacific Southwest Region, Vallejo, CA 94592, United States
[b] Department of Environmental Science and Policy, University of California, Davis, CA 95616, United States
[c] Department of Plant Sciences, University of California, Davis, CA 95616, United States
[d] USDA Forest Service, Plumas National Forest, Quincy, CA 95971, United States
[e] USDA Forest Service, Sierra National Forest, Clovis, CA 93611, United States

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ABSTRACT

We assessed the effectiveness of forest fuel thinning projects that explicitly removed surface and ladder fuels (all but one were combined mechanical and prescribed fire/pile burn prescriptions) in reducing fire severity and tree mortality in 12 forest fires that burned in eastern and southern California between 2005 and 2011. All treatments and fires occurred in yellow pine or mixed conifer forests, in a variety of landscape conditions. Most fires burned under warm, dry conditions, with moderate to high winds. With few exceptions, fire severity measures (bole char height, scorch and torch height, scorch and torch percentage) and tree mortality were much lower in forest stands treated for fuels than in neighboring untreated stands. Fire-tolerant species like *Pinus jeffreyi* and *Pinus ponderosa* exhibited much higher postfire survivorship than fire-intolerant species like *Abies concolor.* Among variables related to fire weather, fuel loading, and treatment age, ten-hour fuel moisture was found to be a better predictor of tree survival in untreated stands than in treated stands, while fuel loading was a better predictor of survival in treated stands. We did not find an effect of treatment age, but our oldest treatments (nine years when burned) were below the mean pre-Euroamerican settlement fire return interval for these forest types. Within treatments, fire severity decreased with distance from the treatment boundary, and canopy fires were almost always reduced to surface fires within 70 m of entering the treatment. In California yellow pine and mixed conifer forests, treatment prescriptions should allow for levels of fire-driven canopy tree mortality (c. 5–15%) that better mimic natural fires. Our results add significantly to the growing evidence that fuel treatments that include removal of surface and ladder fuels in these forest types are highly effective management tools for reducing fire severity and canopy tree mortality. In our opinion, quantitative assessments of fuel treatment effects on fire severity in frequent-fire forest types hardly merit further effort. Rather, we suggest that future work focus on documenting and comparing other ecological outcomes of fuel treatments in burned and unburned forest, such as effects on plant and animal diversity, soil conditions, and habitat heterogeneity.

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1. Introduction

Before Euroamerican settlement began in earnest in the mid-19th century, most drier mixed conifer and yellow pine forests in the western United States supported regimes of relatively frequent, mostly low severity fires. Fires tended to be relatively small in area, and patches of fire-killed trees were rarely more than a few hectares in size. The occurrence of frequent, low and mixed severity fires in these forest types led to ecosystems dominated by a suite of fire tolerant tree species (most notably the yellow pines *Pinus ponderosa* and *Pinus jeffreyi*, but locally including other species like *Quercus kelloggii*, *Pinus lambertiana* or *Populus tremuloides*), and characterized by low fuel loadings, low tree densities, and highly heterogeneous horizontal and vertical structure. In areas of more heterogeneous terrain and topoclimate, tree composition was typically more varied and often included patches dominated by relatively fire intolerant tree species (e.g., species from the genus *Abies*) (Agee, 1993; Sugihara et al., 2006; North et al., 2009; Evans et al., 2011).

Since the mid-19th century, human management practices have fundamentally changed the structure, biota, and ecological processes in mixed conifer and yellow pine forests. These practices include fire exclusion, logging, and grazing, and are compounded by other anthropogenic stressors like air pollution and global warming (Sugihara et al., 2006; Barbour et al., 2007). Yellow pine and mixed conifer forests have become structurally more homogeneous due to the harvest of larger trees and fire exclusion, which has allowed the infilling of dense stocks of younger trees. The

* Corresponding author at: USDA Forest Service, Pacific Southwest Region, Vallejo, CA 94592, United States. Tel.: +1 707 562 8934.
E-mail address: hughsafford@fs.fed.us (H.D. Safford).

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balance of plant species has changed markedly, with once dominant fire tolerant-shade intolerant tree species giving way to shade tolerant trees, and once diverse understory floras reduced to a smaller subset of shade tolerant species. In the absence of fire, and with the local addition of logging detritus, forest fuel loadings have increased by orders of magnitude (Sugihara et al., 2006; Barbour et al., 2007).

Before Euroamerican settlement, fire was very frequent in most mixed conifer and yellow pine forests. In California, presettlement fire return intervals in such forests averaged 11 or 12 years (median 7–9 years), with a range of 5–50 years (Van de Water and Safford, 2011). The presettlement fire rotation for such forests in California (the number of years it took to burn an area equivalent to the overall forest area) has been estimated at 22–27 years (Stephens et al., 2007). Between the 1850s and the early 20th century easily controllable (often naturally ignited) fires were often put out by settlers, but at the same time a rash of large, severe human-caused fires occurred throughout the Sierra Nevada, driven by accumulated logging slash, effects of grazing and logging on forest structure, and rampant (mis)use of fire by loggers, shepherds, miners, hunters, and the like (Sudworth, 1900; Leiberg, 1902).

After the institution of fire suppression in the early 20th century (and the later development of "Smokey-the-Bear" educational efforts), fire became a comparatively rare phenomenon in California forests, reaching its nadir in the 1960s to early 1980s, when fire rotations in the Sierra Nevada and NW California ranged from 600–900+ years (Miller et al., 2009, 2011). Since the mid-1980s however, drier mixed conifer and yellow pine forests across the western United States have experienced a notable increase in fire frequency, mean and maximum size, and annual burned area. Some regions have experienced proportional increases in fire severity and some have not, but the overall area of high severity ("stand-replacing") fire in these forest types is also rising (Miller et al., 2009, 2011; Dillon et al., 2011). In concert with these fire trends, numbers of fire-caused human fatalities and destroyed and damaged homes are also mounting. This reversal in fire trends comes even as federal and state governments deploy tens-of-thousands of fire fighters and spend billions of dollars annually on fire management (Stephens and Ruth, 2005). Scientific investigations of these trends have identified climate warming and increasing forest fuels as the chief culprits, with the balance between the two depending on such factors as geography, regional climate, forest type, management history, and human contributions to ignition density (Schoennagel et al., 2004; Noss et al., 2006). Current empirical data, patterns in the paleo-record during similar warming periods, and future modeling all suggest these tendencies toward increasing fire activity and impact will continue and perhaps accelerate, as both temperatures and fuel loads continue to increase (e.g., Miller and Urban, 1999; Whitlock et al., 2003; Lenihan et al., 2003; Westerling et al., 2006; Westerling and Bryant, 2008; Miller et al., 2009; Gedalof, 2011; National Research Council, 2011).

These trends have led federal and state fire and resource agencies to prioritize management actions meant to reduce fire spread and fire severity, as well as to protect human infrastructure and sensitive biological habitat. The principal approach employed in the western US is the reduction of forest fuel loadings in areas that are near important human, biological, or environmental assets, or in strategic locations that might allow more rapid control of a developing fire. Such treatments are also being used more and more frequently in efforts to restore forest structures and biological habitat and to increase resilience to drought, fire and other disturbances (Reinhardt et al., 2008; Schoennagel and Nelson, 2011). Fuel reduction in mixed conifer and yellow pine forests in the western US typically involves some combination of mechanical, hand, and/or burning treatments. Studies comparing different fuel treatment techniques have generally found that the most effective treatments incorporate targeted mechanical or hand removal of larger fuels followed by a burning treatment to reduce surface fuels (Schwilk et al., 2009; Evans et al., 2011).

In this contribution, we report results from a large ongoing study of fuel treatment effectiveness and ecological effects in mixed conifer and yellow pine forests in the California National Forests. The principal purposes of our monitoring program are (1) to assess the effectiveness of completed forest fuel treatments (by "completed" we mean treatments which explicitly remove surface fuels from the treatment site, usually by fire) in modifying fire behavior and ameliorating fire severity, and (2) to evaluate some of the ecological effects of these treatments, both before and after fire. In an earlier publication (Safford et al., 2009), we reported on the outcomes of fire in forest fuel treatments that were burned in the 2007 Angora Fire at Lake Tahoe, California. Here we report on the effectiveness of fuel treatments in eleven additional fires that occurred in mixed conifer and yellow pine forest types across eight National Forests in California.

2. Methods

2.1. Study sites

In addition to the Angora Fire, we selected 11 additional fires for post-fire sampling in treated areas and adjacent untreated areas, to record data on fire severity and tree mortality. The locations of our study sites are shown in Fig. 1 and their geographic coordinates are given in Table 1. Sites were chosen after field visits and discussions with National Forest staff, including fire fighting personnel assigned to the respective fires. We sampled only fires that burned in yellow pine or mixed conifer forest (see below), and only sites within fires where forest thinning had included an explicit treatment of surface and ladder fuels (the Dawson Springs treatment within the American River Complex is the sole exception to this rule). Elevations of our sample sites ranged from 1250 m to 2235 m (Tables 2a and 2b). Climate across our study area is Mediterranean-type (cool, wet winters and warm, dry summers), with somewhat of an interior continental influence (colder winters, and summers with greater chance of episodic precipitation) at sites east of the Cascade-Sierra Nevada crest. Mean annual temperatures at our sites range from 7.2 to 11°, and mean annual precipitation ranges from c. 370 mm to 1700 mm (Table 1).

The fires we sampled were of varying sizes (112–15059 ha), and burned under varying conditions (Tables 1 and 2a). Seven fires were ignited by humans, five were ignited by lightning. Most of the fires we sampled burned under dry, relatively warm conditions, with moderate to high winds (Tables 2a and 2b). Most fires occurred under conditions where the Energy Release Component (ERC; a measure of potential energy release by the fire) was above the 85th percentile for the day of the year the treatment area burned (mean = 85th percentile, median = 90th percentile); three fires burned under relatively low ERC conditions (Tables 2a and 2b). Weather and fuel conditions varied substantially among fires, with 10-h fuel moistures ranging from 3.7 to 8%, minimum relative humidity from 8 to 32%, and maximum temperatures from 16 to 33 °C (Tables 2a and 2b).

2.2. Field methods

We carried out field sampling between July 2007 and September 2011, visiting each site as soon after fire as practical, and repeating visits once each summer. The first year of sampling for each fire is noted in Table 1. We sampled sites previously treated for fuels within the fire perimeter, plus areas immediately adjacent

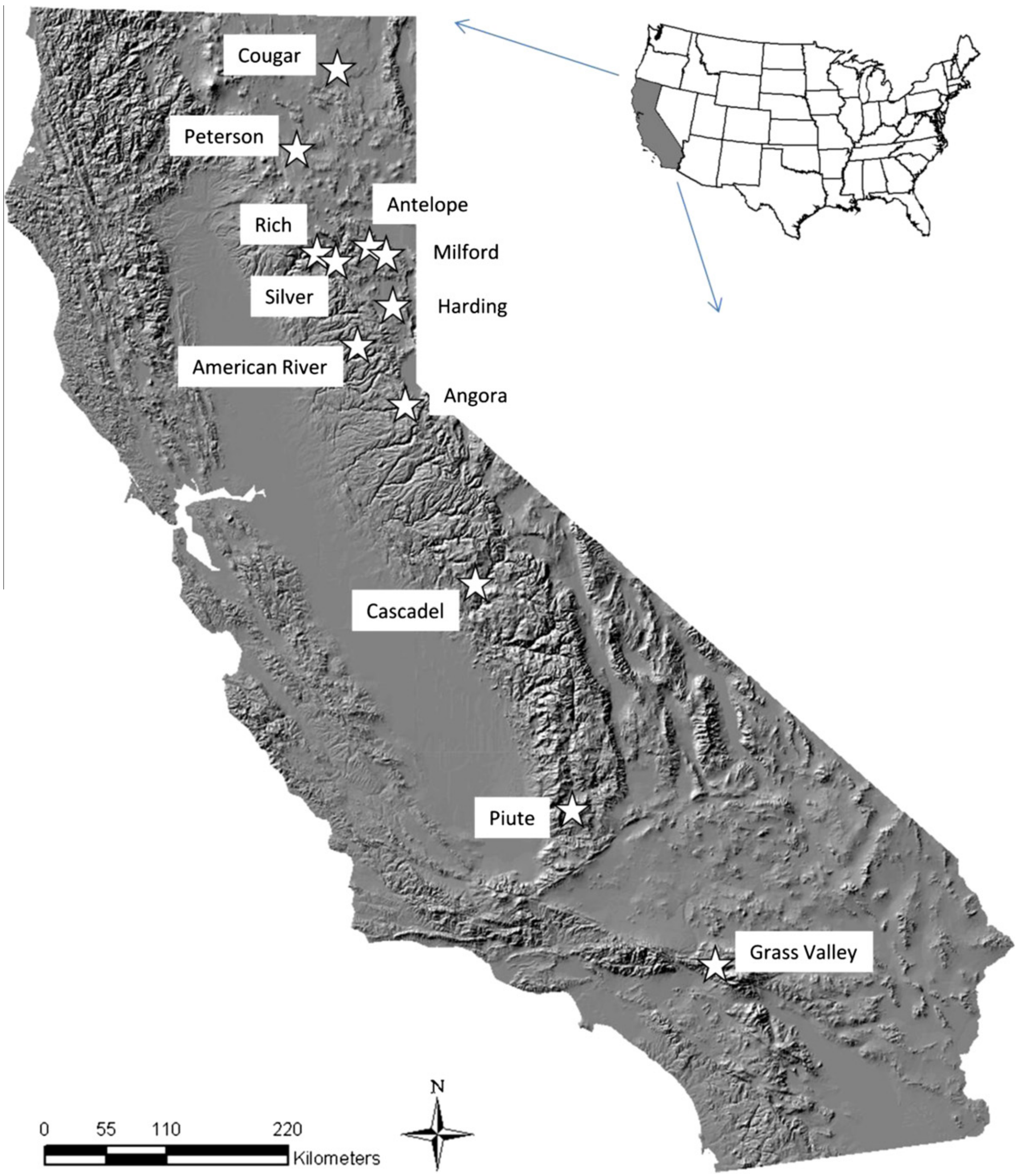

Fig. 1. Locations of the 12 California fires included in this study.

to these sites in untreated forest, using linear transects that were almost always aligned perpendicular to fuel treatment boundaries (Fig. 2). We sampled fuel treatments where fire spread from adjacent untreated National Forest land into the treated area. In most cases we selected approximate transect locations before visiting the field by using GIS data that showed treatment boundaries. Final transect placement was determined on the ground by using maps and field validation to determine the length of the treatment boundary, and positioning the transect perpendicular to the boundary at a randomly determined point along the boundary length. We also positioned transects to avoid areas that had been harvested after fire, roads, riparian areas, locations where intensive fire fighting activities had occurred, and areas where there were major slope differences between treated and untreated stands (see Tables 2a and 2b). In a few cases we were not able to avoid road boundaries between treated and untreated stands. These are identified in Tables 2a and 2b. We obtained estimates of surface fuels (sum of 1 h to 1000 h+) for seven treated areas and ten untreated areas. Data (from field measurements and/or photo series estimates) were acquired from National Forest staff where possible. In a few cases, we were able to make direct measurements of surface fuels in the field where similar treatments had been recently completed in unburned forest. Where both data sources were available, we averaged the two.

Our transect sampling protocol was designed to allow the measurement of linear changes in fire effects as fire passed from untreated to treated stands, but also to minimize the influence of weather variability in confounding the interpretation of those effects. Under the fire spread rates which characterized the fires we sampled (1–2 km/hr), most of the transects we sampled would

Table 1
General information for the 12 sampled fires.

Fire name	National forest	Ignition date	Cause[a]	Size (ha)	Lat.[b]	Long.[b]	Mean annual ppt. (mm)[c]	Mean annual temp. (°)[c]	Jan. mean min. temp. (°)[c]	July mean max. temp. (°)[c]	First sample year[d]
American River Complex	Tahoe	21-Jun-08	L	8190	39.211°	120.588°	1700	10.2	−1.2	26.7	2009
Angora	Lake Tahoe Basin	24-Jun-07	H	1243	38.887°	120.039°	974	6.3	−7.8	25.9	2007/2008
Antelope Complex	Plumas	5-Jul-07	L	9004	40.14°	120.582°	814	7.4	−6.5	26.9	2009
Cascadel	Sierra	11-Sep-08	H	112	37.249°	119.444°	1065	10.8	−0.9	27.7	2010
Cougar	Modoc	8-Jun-11	L	716	41.65°	121.43°	392	8.8	−4.9	28.5	2011
Grass Valley	San Bernardino	22-Oct-07	H	501	34.265°	117.187°	697	11.0	−1.5	27.8	2010
Harding	Tahoe[e]	24-Aug-05	H	914	39.635°	120.314°	641	7.2	−6.8	26.4	2010
Milford Grade	Plumas	4-Apr-09	L	131	40.109°	120.389°	669	7.5	−5.7	26.5	2009/2010
Peterson	Lassen	21-Jun-08	L	3235	40.917°	121.335°	559	9.6	−6.0	30.7	2009
Piute	Sequoia	28-Jun-08	H	15059	35.502°	118.337°	369	8.1	−4.5	25.5	2009
Rich	Plumas	29-Jul-08	H	2464	40.041°	121.135°	1099	10.5	−1.2	28.2	2009
Silver	Plumas	19-Sep-09	H	125	39.949°	121.09°	1321	9.5	−2.1	26.5	2010

[a] L – lightning, H – human.
[b] Of the center of the fire.
[c] 1970–2004 data, from the PRISM Climate Group, Oregon State University, http://prism.oregonstate.edu, created 4 Feb 2004.
[d] Where there are two years listed, the first year pertains to measures of severity, the second year to mortality.
[e] Treatments were on California Department of Fish and Game land adjoining the Tahoe National Forest.

have been completely passed by the flaming front in a period of <15 min. Transects that were more parallel to the direction of fire spread would have taken somewhat longer to burn than transects perpendicular to fire spread.

We sampled from 10–28 points per transect, situated about 20 m apart (with the caveat that points were moved if trees would have been resampled at adjacent points), with approximately the same number of points within treated and untreated areas. We used a sampling protocol derived partly from the point-center quarter method (Cottam and Curtis, 1956), taking measurements of the nearest tree $\geqslant$10 cm (3.94 inch) diameter at breast height (dbh) in each of the four compass quadrants radiating from our sampling point. At each point we measured the slope and took a GPS reading. We measured the distance to each chosen tree using a hand-held laser range finder, identified the tree to species and measured the dbh, overall height and height to the base of the crown. Crown base height was measured as the vertical distance from ground level to the lowest whorl with live branches in at least two of four quadrants around the stem (Helms, 1998); estimates of the prefire live crown-base height were necessary where trees were severely burned. We also coded trees as dead or alive: if green needles were seen on the tree, we scored it "alive", if needles were completely scorched (browned) or torched (consumed) we scored it "dead". In this paper, we only analyze tree survivorship data collected at least one full year after fire (the one exception being the Cougar Fire), as immediate postfire assessments of survivorship are likely to underestimate fire-caused mortality (see Table 1 to compare date of fire with date of sampling). All sampling was conducted many weeks after bud-break.

To gauge fire effects, during our first visit to a site we measured a number of standard fire severity metrics for each sampled tree (see Agee, 1993). We used a laser rangefinder to measure bole char height (height of surface flame effects on the main tree trunk), scorch height, and torch (consumption) height. We also estimated percent crown scorch and percent crown torch, by making an ocular determination of the percentage of the tree canopy had been consumed by fire (torch percentage) and consumed and browned (scorch percentage); torch percentage is thus a component of scorch percentage. At three sites our first sampling visit was more than two years after fire (see Table 1). In these cases, we were not able to measure variables related to crown scorching or torching due to needlecast. A more detailed description of these field data collection procedures is presented in Safford et al. (2009).

2.3. Data analysis

We compared fuel loading between treated and untreated stands (data were not significantly different from normal) using a one-sided two-sample *t*-test under the assumption of unequal variances.

We analyzed the effect of fuel treatments on measures of fire severity using five separate measures: height of bole char, height of crown scorch (where leaves had been affected by fire), height of crown torch (where leaves had been incinerated by fire), and percentage of total crown both scorched and torched. We used the arcsin square-root transformation to normalize the variables that were recorded as percentages. For each fire, we tested whether the means of these five response variables were different in treated and adjacent untreated parts of the forest, using linear mixed models. In these models, the response variable was the severity measure, the fixed effect was Treatment (1/0) and we included random effects for transect nested within fire to reflect the grouped structure of the data. We compared the percentage of canopy mortality between treated and untreated areas using the same analytical approach, by averaging the four quarter measures of tree survivorship at each point into a single "percent survivorship" measurement at each center point along a transect.

We also examined how fire severity and tree survivorship were affected by distance from the treatment boundary, i.e., the edge between treated and untreated areas. To do this, we aggregated the measurements taken at each point along the transect. For consistency, we restricted this analysis of distance effects to the five closest sample points on either side of the treatment boundary, because this was the standard transect length for all fires except for Angora (variable transect lengths) and Peterson (15 points on each transect). For both the treated and untreated halves of the transect, we modeled the same six response variables described above, as a linear function of relative position to the treatment

Table 2a
General information from the sampled fuel treatments, including number of treatments and sampling points, treatment completion date, date burned, geographic location, and dominant tree species.

Fire	No. of transects	No. of sample points[a]	Treatment name	Type[b]	Completion date	Date burned by wildfire	Mean elev. (m)	Mean Lat.	Mean Long.	Dominant tree spp.[c]
American River Complex*	3	30	Texas Hill/ Dawson Spring	1,2	1999?	30-Jun/1-Jul-08	1700	39.23	120.63	PIPO, ABCO, PILA, CADE
Angora	8	111	Various	3	2005–2007	24-Jun-07	1970	38.89	120.04	PIJE, ABCO
Antelope Complex*	3	30	Antelope Border DFPZ	4,5	2006	7-Jul-07	1700	40.14	120.57	PIJE, PIPO, ABCO
Cascadel	1	10	Whiskey	9	2002	12-Sep-08	1710	37.25	119.44	ABCO, PIPO, PILA, CADE
Cougar*	2	20		5	2004	8-Jun-11	1420	41.63	121.42	PIPO
Grass Valley	3	30	Tunnel 2	8	2005–2006	22-Oct-07	1677	34.26	117.23	QUKE, ABCO, QUCH, PIJE, PIPO
Harding	3	30	Antelope Valley	5	2001	26-Aug-05	1675	39.63	120.31	PIJE, ABCO, JUCA
Milford Grade	2	20	Last Chance	4	2005	22-Apr-09	1720	40.11	120.40	PIJE, ABCO
Peterson*	3	45	Pittville	6,7	2006	23/24-Jun-08	1320	40.89	121.32	PIJE, ABCO, JUCA
Piute	3	33	Kelso	5	1999	8/9-Jul-08	2235	35.48	118.36	PIJE, ABCO
Rich	3	30	Kingsbury-Rush	4,5	2005	29-Jul-08	1878	40.06	121.13	PIJE, ABCO, PILA, CADE
Silver	3	30	Meadow Valley	4	2004	19-Sep-09	1250	39.95	121.09	PIPO, ABCO, CADE, PILA

* In these fires, a road formed the boundary between treated and untreated stands in at least some of the sampled treatments.
[a] Sample points were split approximately equally between treated and untreated stands.
[b] See following codes for treatment types: (1) commercial thin + precommercial thin + unknown; (2) commercial thin (whole tree yarding); (3) commercial thin + precommercial thin + hand pile + pile burn; (4) precommercial thin + hand pile + underburn; (5) commercial thin (whole tree) + underburn; (6) commercial thin + precommercial thin + underburn; (7) precommercial thin; (8) salvage harvest + precommercial thin + chipping + underburn; (9) commercial thin + machine pile + pile burn.
[c] ABCO = *Abies concolor*, CADE = *Calocedrus decurrens*, JUCA = *Juniperus californica*, PIJE = *Pinus jeffreyi*, PILA = *Pinus lambertiana*, PIPO = *Pinus ponderosa*, QUCH = *Quercus chrysolepis*, QUKE = *Quercus kelloggii*.

Table 2b
General information from sampled fuel treatments, continued: measures of fuel loading, fire weather and fire danger, and published and unpublished reference sources with further information.

Fire	Fuel loading[a]		Fire weather/fire danger measures					References
	Treated fuel load (t/ha)	Untreated fuel load (t/ha)	ERC (%ile)[b,c]	10-h fuel moisture (%)[c]	Max. temp. (°C)[c]	Min. rel. Humidity (%)[c]	Wind speed (gust) (km/hr)[c]	
American River Complex		49.1	81/81 (89/89)	5.2/5.3	26	16	0–8 (8)	Safford, 2008
Angora	11.8	57.9	44 (90)	5	23	11	15–30 (65)	Murphy et al., 2007; Safford et al., 2009
Antelope Complex	18.7	48.5	90 (87)	3.7	33	19	10–35 (>60)	Fites et al., 2007; Murphy et al., 2010
Cascadel		58.4	60 (90)	6.4	28	27	0–16 (25)	
Cougar	4.7	10.0	46 (>90)	8	20	32	0–5 (13)	USFS, 2011
Grass Valley			70 (64)	5.2	16	8	15–30 (65)	Rogers et al., 2008
Harding		18.2	51 (62)	6.8	29	18	0–5 (20)	
Milford Grade	8.7	15.5	55 (90)	8.0	22	14	4–30 (53)	USFS, 2010a; Murphy et al., 2010
Peterson			72/71 (91/90)	7.4/7.8	24	17	5–17 (35)	Merriam, 2008; Murphy et al., 2010
Piute	7.9	25.0	97/86 (90/85)	5.8/6.0	29	7	0–11 (37)	Meyer and Safford, 2010
Rich	26.8	56.0	82 (87)	5	32	15	6–10 (30)	USFS, 2009; Murphy et al., 2010
Silver	29.7	36.5	84 (92)	6.5	31	14	9–16 (30)	USFS, 2010b; Murphy et al., 2010

[a] Tons/ha calculated as sum of all surface fuels (1–1000 h+). Mean of all measures available for sampled treatments and adjacent untreated stands, data obtained from National Forest staff (field measurements and/or photo series estimates) and/or direct field measurement by our crews in unburned sites. Where both data sources were available, we averaged the two.
[b] Energy Release Component, from the National Fire Danger Rating system. ERC is the 24-h, potential worst case, total available energy (BTUs) per unit area (in feet^2) within the flaming front at the head of a fire. "% ile" is percent of times over a period of many years that ERC falls below (i.e. fire danger is lower than) the measurement given for the given day.
[c] Calculated from the nearest weather station at similar elevation for the day the fuel treatments in question burned. ERC and fuel moisture data courtesy of Larry Hood, US Forest Service Region 5.

boundary: relative positions of -5 to -1 in the untreated half, and 1 to 5 in the treated half.

In addition to looking at severity patterns in individual fires, we assessed the overall effect of fuel treatments on fire severity across all 12 fires using mixed–effects models to analyze the combined data. The response variables in these regressions were, for each individual tree: height of bole char, height of crown scorch, height of crown torch, and mortality. As described above, each individual tree was associated with a sample point along a transect within a fire, so to account for this grouping structure and the resulting

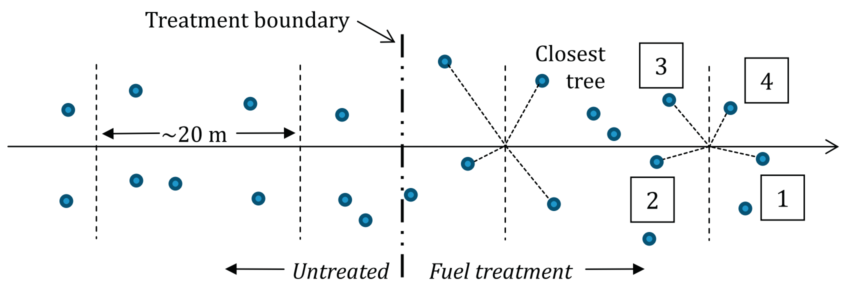

Fig. 2. Transect configuration. Nearest trees were sampled in each compass quadrant (1–4).

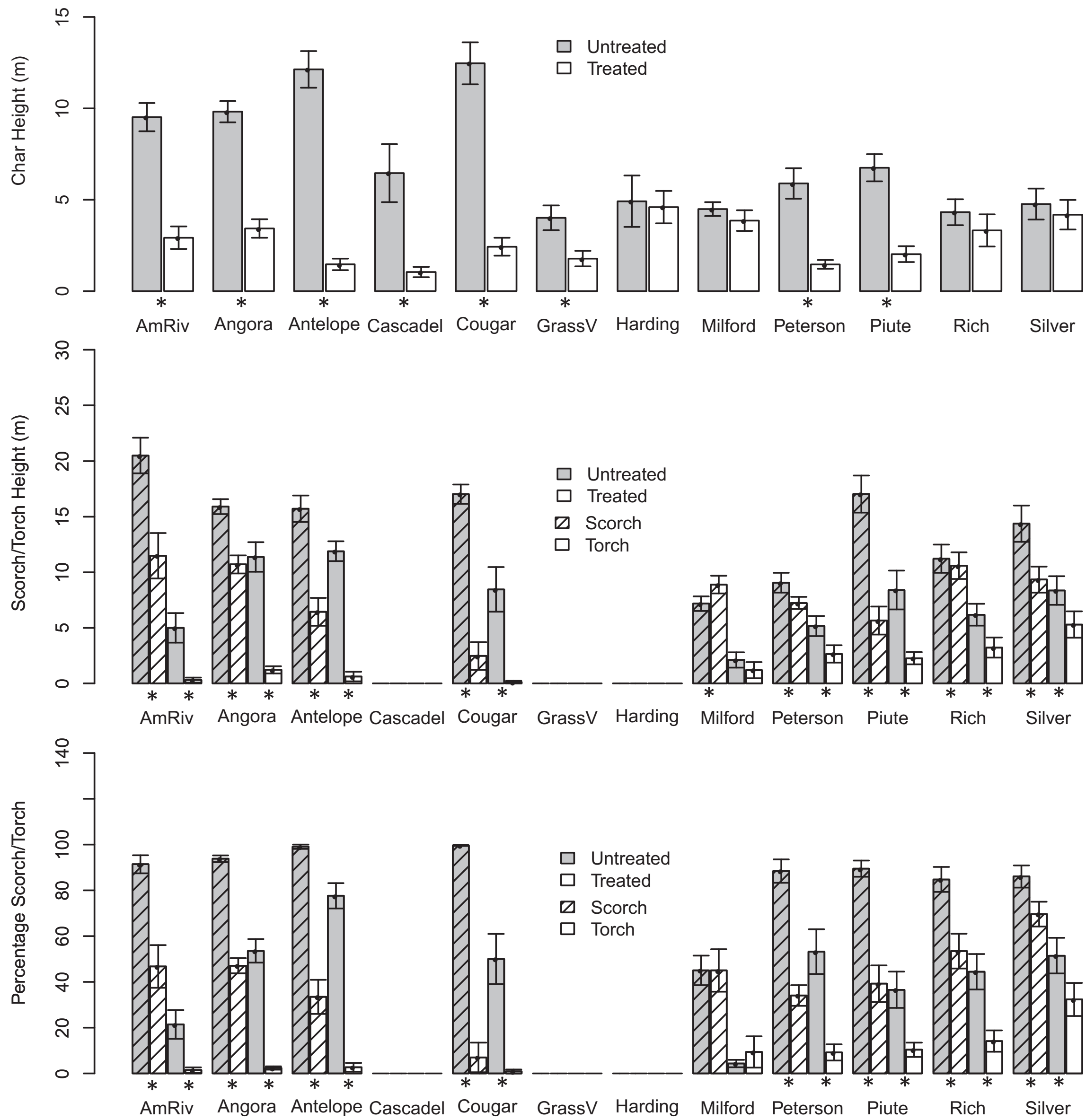

Fig. 3. Fire severity response to fuel treatment, in terms of (a) char height, (b) scorch and torch height, and (c) scorch and torch percentage. Shaded bars represent untreated sites and white bars represent treated sites; hatched bars represent scorch and empty bars represent torch. All data were pooled from multiple transects except Cascadel, which only had one transect. Bars represent ± one standard error; asterisks denote significant ($P < 0.05$) differences between treated and untreated means within sites. *Crown* scorch and torch% were not assessed for Cascadel, Grass Valley or Harding, because two years or more had elapsed since the fire when measurements were taken and valid measures of scorch and torch could no longer be made due to needle fall.

non-independence among trees, we included random effects for sample point nested within transect nested within fire (Gelman and Hill, 2007; McMahon and Diez, 2007). We expected that tree species might respond differently to fire, so we also included random intercepts for species. In addition to assessing the overall effect of fuel treatment on fire severity, we also used these regressions to compare the influence of treatment versus topography (slope steepness and insolation), pre-burn tree density, and tree

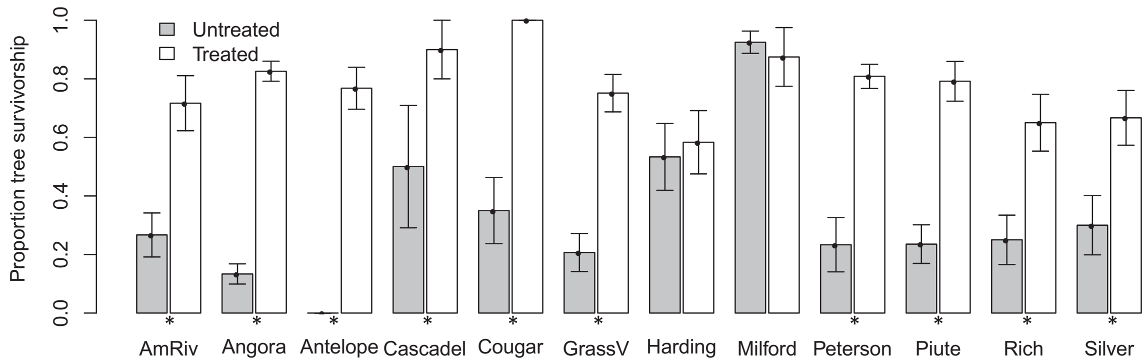

Fig. 4. Tree survivorship during the first year measured after fire (See Tables 2a and 2b), in response to fuel treatment. Shaded bars represent untreated sites and white bars represent treated sites. At all sites except Cascadel, data were pooled from multiple transects. Bars represent ± one standard error; asterisks denote significant ($P < 0.05$) differences between treated and untreated means within sites.

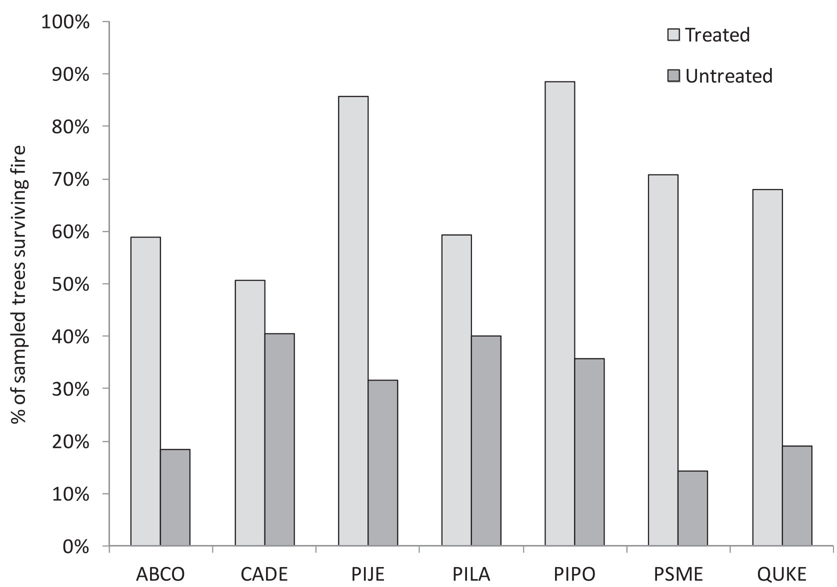

Fig. 5. Percent of sampled trees surviving fire in treated and untreated stands for species occurring in at least four fires and with $N \geqslant 20$ in both treated and untreated samples. All assessments of survival made at least one year after fire, except for Cougar (4 months postfire). Species codes as in Tables 2a and 2b; PSME = *Pseudotsuga menziesii.*

Table 3
Linear regression coefficients and associated *P*-values for six severity response measures as a function of position relative to the treatment boundary. Positions were generally 20 m apart (see Section 2). Untreated coefficients represent effect of moving from the untreated forest toward the treatment boundary, and treated coefficients represent effect of moving away from the treatment boundary into treated forest.

Variable	Untreated coefficient	*P*-value	Treated coefficient	*P*-value
Char Ht	−0.232	0.330	−0.516	0.003
Scorch Ht	−0.117	0.751	−0.698	0.048
Torch Ht	−0.155	0.686	−0.452	0.039
Scorch Pct	−0.008	0.521	−0.085	<0.001
Torch Pct	−0.017	0.421	−0.032	0.002
Survivorship	0.054	0.005	0.079	<0.001

size, as well as interactions among these factors. The fixed effects in each model were thus fuel treatment (1/0), slope, mean distance from the sample point to the nearest tree in each compass quadrant, and tree size (log-transformed diameter at breast height – DBH).

For each response variable, we started with a "full model" that included four main effects (treatment, slope, mean tree distance, insolation) and all two-way interactions among the fixed effects. We then removed the model term with the least significant coefficient (using Wald Z-score for comparison) and compared the reduced model to the full model using the Akaike Information Criterion (AIC) (Burnham and Anderson, 2002). If the simpler model had the same or lower AIC score, we removed the variable. We repeated this variable-dropping procedure until no variables could be removed without worsening the AIC score. All models were fitted in R (R Development Core Team, 2005) using the lme4 library (Bates and Maechler, 2010).

For some potentially important driving variables, including weather at the time of fire, fuel conditions, and fuel treatment age (the number of years between thinning and fire) we have summary information for each fire (e.g., only one value of each variable for each fire; Tables 2a and 2b). We assessed the influence of these factors on fire severity using the mean severity values for each fire × treatment combination. Thus, for each fire we calculated mean severity in treated areas and mean severity in untreated areas. For weather and fuel moisture data, we then regressed the mean severity and survivorship data on the interaction between treatment and each explanatory variable: ERC, ERC percentile, maximum temperature, minimum relative humidity, average wind speed, wind gust speed, and ten-hour fuel moisture. Treatment age only applies to treated areas, so in this case we carried out a simple linear regression against the mortality/survival variables for

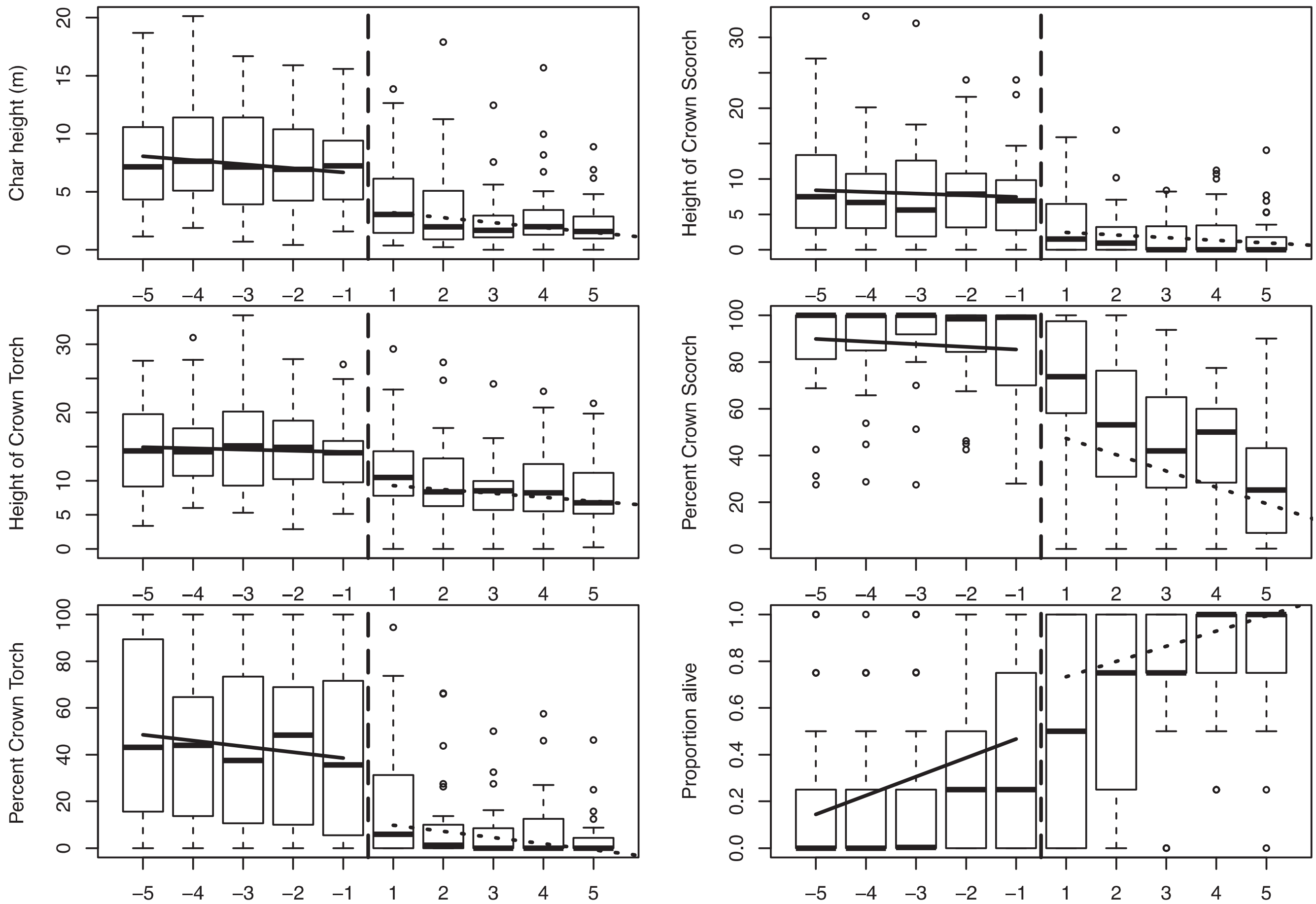

Fig. 6. Change in fire severity with position relative to treatment boundaries, across all 12 fires. Position refers to the order of plots away from the boundary, which is position 0. Negative numbers are untreated points, positive numbers are treated points, and the dashed vertical line indicates the treatment boundary. Individual points on boxplots are plots that exceeded the third quartile of the data plus the inter-quartile range. Plots were usually spaced 20 m apart. Solid (untreated) and dashed (treated) lines represent best-fit regression lines as a function of position. See Table 3 for regression coefficients and measures of significance.

treated areas. We had fuel loading data for untreated areas at ten of the fires and for treated areas at seven of the fires. In this case, we carried out simple linear regression between the mortality/survival data and the fuel loading values for the treated sites alone, and the untreated sites alone.

3. Results

3.1. Fuel loading

Surface fuel loadings averaged 37.51 tons ha^{-1} (SE = 3.70) in untreated stands and 15.47 tons ha^{-1} (SE = 5.98) in treated stands. These were significantly different at $P < 0.004$ ($t = 3.134$). Where we had fuel loadings for both treated and untreated stands (see Tables 2a and 2b), the mean difference was 20.16 tons ha^{-1} (SE = 5.84).

3.2. Fuel treatment effects on fire severity

Measures of fire severity – char height, and height and percent of crown torch and scorch – were significantly reduced in treated forest compared to untreated forest at most sites. Across the 12 sampled fires, bole char heights were always lower in treated than in untreated areas; differences were statistically significant at eight of the 12 sites (Fig. 3a). Among sites where fuel treatments had a significant effect, the average reduction of char height in treated areas across all sites was 5.8 m. Significant reductions in char height ranged from 4.4 m at Peterson, to 10.7 m at Antelope.

The height and percentage of both crown scorch and torch generally decreased in treated areas, for transects in which those responses were measurable (9 of 12 sites; Fig. 3b and c); eight of nine fires showed higher torch height in untreated vs. treated forest, and eight of nine showed higher scorch height in untreated forest as well. Similar patterns characterized scorch and torch percentage. The one consistent exception was the Milford Fire, which had one of the lowest energy release components of all fires (Tables 2a and 2b), and had some of the lowest severity measures in the untreated forest of any site. Among sites where fuel treatments significantly reduced scorch and torch percentage, crown scorch decreased by an average of 45.6%, and crown torch decreased by an average of 36.3%. Significant reductions of crown scorch in treated areas ranged from 16% (Silver) to 92% (Cougar). Significant reductions of crown torch in treated areas ranged from 19% (American River) to 75% (Antelope).

3.3. Fuel treatment effects on tree survivorship

Concordant with the decrease in fire severity in treated forests, adult tree survivorship was significantly increased in treated areas compared to untreated areas (Fig. 4), for all sites except Milford and Harding. Significant increases in survivorship in treated areas ranged from 45% at American River, to 77% at Antelope, where untreated areas experienced complete canopy mortality. Species identity had a significant effect on survival probability. We encountered 12 tree species in our sampling, nine conifers and three hardwoods (all *Quercus* spp.). Seven species were sampled in more than three fires, and had >20 trees in both treated and untreated samples (Fig. 5). *Pinus jeffreyi* ($N = 527$) and *P. ponderosa* ($N = 364$) exhibited the highest survival rates (>63% overall survival, >86% survival in treated forest); *Abies concolor*, *Pseudotsuga menziesii*, and *Quercus kelloggii* ($N = 360$, 66, and 70, respectively) showed the lowest overall survival (all <40%) and the lowest survival in untreated forest stands (all ⩽19%). *Calocedrus decurrens* (=*Libocedrus decurrens*; $N = 133$) exhibited the lowest survival in treatments (51%) and the smallest difference between treated and untreated stands; *P. menziesii* showed the largest difference in survival between treated and untreated stands (Fig. 5).

3.4. Distance effects

Fire severity showed marked decreases in the treated areas at plots further away from the treatment boundaries, particularly within about 3 positions (40–70 m) of the boundary (Table 3, Fig. 6). Interestingly, the increasing trend in tree mortality from untreated to treated forest began even before the treatment boundary was reached (Table 3).

3.5. Overall severity patterns

When data from all the fires were combined, treatment had a very strong negative effect on all severity measures – for all models, the regression coefficient for treatment was the largest in magnitude, always associated with lower fire severity, and always highly significant (Table 4). The size of the tree was also always important: log(DBH) had a significant and substantial effect on all severity measures (Table 4). Larger trees tended to have higher absolute scorch and torch heights simply as a function of being taller with higher canopies. Larger trees also had a significantly higher chance of survival. There was a significant interaction term in three out of the four models for treatment and log(DBH): in treated areas, tree size had a weaker effect on char height, scorch height, and torch height (Table 4).

For char height, crown torch and tree survival, there was a significant interaction between slope and treatment, indicating slope had a larger effect on fire severity in treated areas (Table 4). In treated areas, steeper slopes tended to have more severe fire effects and lower tree survival. In untreated areas, in contrast, slope had a weaker association with fire severity or tree survival for most severity measures. In the case of char height, the main effect of slope was also significant, but smaller and with the opposite sign to that of the interaction term (main effect of slope = −0.20; interaction with treatment = 0.45) (Table 4). In untreated stands, the effect of slope is the sum of the main effect and interaction term, so steeper slopes were still associated with higher char heights in untreated areas, but less strongly than in treated areas.

Pre-fire tree density, expressed as mean distance from plot center to the closest tree in each quadrant, had a positive association with bole char height and a negative association with crown torch height, though these effects were relatively small (Table 4). These effects of tree spacing were small because most of the variation in tree density was accounted for by the fixed effect for thinning treatment. Accordingly, if we remove fuel treatment from the model, higher tree density is significantly associated with higher bole char, crown torch, and tree mortality (results not shown). Random effects indicated substantial residual variation among points within transects and among transects within fires. There was much less variation, in contrast, among fires and among tree species.

We also tested the significance of large-scale variables on mean fire severity across entire fires. Of the weather and fuel condition variables, we only found two significant relationships. Ten-hour fuel moisture (percent moisture content of wood with diameter 0.64–2.54 cm) had a significant relationship across untreated and treated stands, strongly affecting tree survival ($P < 0.001$). There was a marginally significant trend toward an interaction between treatment and ten-hour fuel moisture ($P = 0.086$), such that treated areas experienced relatively less mortality under very dry conditions (simple regressions gave an r^2 of 0.532 [$P = 0.026$] for untreated stands and 0.371 [$P = 0.081$] for treated stands; Fig. 7).

Table 4
Regression coefficients for fixed effects of fuels treatment and environmental variation on fire severity measures across 12 Sierra Nevada wildfires: (a) log-transformed char height; (b) crown scorch height; (c) crown torch height; and (d) tree survival.

Variable	Coefficient	SE	Wald z-value	*P*
(a) Bole char height				
Treatment	−1.144	0.099	−20.815	<0.001
logDBH	0.263	0.055	10.659	<0.001
Slope	−0.203	0.051	−3.952	<0.001
Tree spacing	0.068	0.031	−2.186	0.037
Treated * logDBH	−0.119	0.033	−3.643	0.005
Treated * slope	0.448	0.054	8.304	<0.001
(b) Crown scorch height				
Treatment	−0.787	0.055	−14.288	<0.001
logDBH	0.445	0.024	18.422	<0.001
Slope	−0.020	0.056	−0.352	0.375
Treated * logDBH	−0.257	0.033	−7.777	<0.001
logDBH * slope	−0.052	0.018	−2.934	0.005
(c) Crown torch height				
Treatment	−0.790	0.064	−12.406	<0.001
logDBH	0.231	0.027	8.569	<0.001
Slope	−0.176	0.067	−2.630	0.013
Tree spacing	−0.171	0.049	−3.459	0.001
Treated * logDBH	−0.159	0.036	−4.355	<0.001
(d) Tree survival				
Treatment	4.710	0.355	13.286	<0.001
Slope	0.018	0.346	0.053	0.958
logDBH	1.177	0.133	8.866	<0.001
Insolation	−0.629	0.265	−2.375	0.018
Treated * slope	−0.769	0.355	−2.168	0.030
Treated * insolation	1.193	0.343	3.481	<0.001

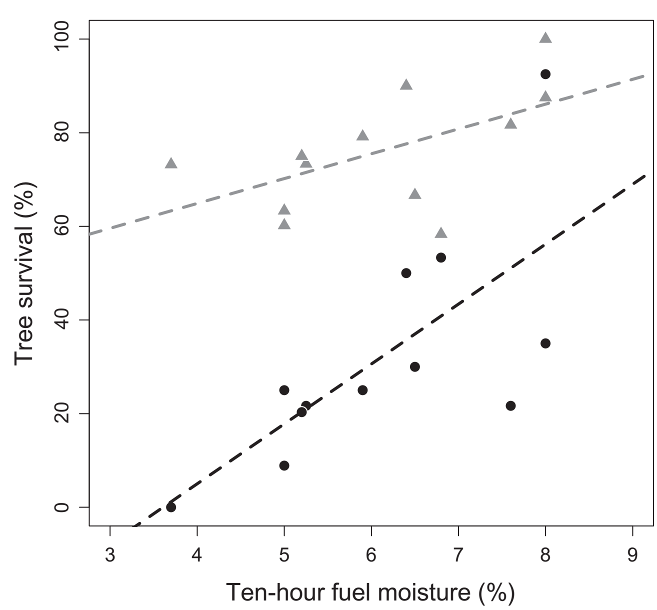

Fig. 7. Average tree survival in each of 12 Sierra wildfires versus 10-h fuel moisture at the time of fire. Black circles are tree survival for each fire averaged across sample locations in untreated forest; gray triangles are tree survival for each fire averaged across sample locations in treated forest. Fit of the regression model was good ($r^2 = 0.78$) for a model including treatment, ten-hour fuel moisture, and their interaction.

Based on the regression lines in Fig. 7, an increase in ten-hour fuel moisture from 4% to 8% corresponds to a c. 30% rise in tree survival in treated stands (65% to 85%). However, the same change in ten-hour fuel moisture corresponds to a 1000% rise in the probability of survival in untreated forest (5% to 55%). Fuel loading (1–1000 + hr fuels) was significantly negatively related to survival in the treated stands ($r^2 = 0.646$, $P = 0.029$), but not in untreated stands ($r^2 = 0.260$, $P = 0.132$).

4. Discussion

Our results corroborate those found in other published statistical assessments of fuel treatment effectiveness (e.g., Pollett and Omi, 2002; Cram et al., 2006; Strom and Fulé, 2007; Martinson and Omi, 2008; Safford et al., 2009; Prichard et al., 2010; Hudak et al. 2011). In most of these cases treatment effects during single fire events were analyzed (Pollett and Omi studied four fires; Cram et al. studied five); here we find similar effects across a sample of 12 wildfires. Until a few years ago, rigorous statistical assessment of effectiveness had only rarely been carried out, but earlier questions about the empirical outcomes of fires in completed fuel treatments can now be put to rest. For yellow pine and mixed conifer forests, the evidence seems overwhelming: with few exceptions, fuel treatments that incorporate explicit removal of surface fuels can be expected to significantly reduce fire severity and canopy tree mortality, even under relatively extreme weather conditions. Indeed, we believe that stand-scale quantitative assessments of fuel treatment effects on wildfire severity in frequent-fire forest types hardly merit further effort. Rather, we suggest that future work focus on the ecological outcomes of fuel treatments in burned and unburned forest, such as for species diversity, understory plants, soil conditions, habitat heterogeneity, and the like.

All but one of the treatments we sampled used prescribed fire in some way (e.g., broadcast burning or burning of piled fuels). Many studies show that prescribed fire in combination with hand or mechanical removal of larger materials is the most effective way to reduce subsequent wildfire severity (Schwilk et al., 2009; Stephens et al., 2009; Evans et al., 2011). Although the theoretical basis for removing surface and ladder fuels has been understood for quite some time (Weatherspoon and Skinner, 1996; Graham et al., 2004; Agee and Skinner, 2005), many management units lack sufficient funding to complete fuel treatments in a reasonable amount of time. Surface fuels are often high after the first treatment entry, and if they are not reduced by some means, they are there to burn in a subsequent wildfire and contribute to increased fire severity effects (e.g., greater tree mortality). A number of the fires we sampled showed this effect. For example, unburned or unremoved fuel piles within parts of the American River, Angora, and Cougar Fires were burned by wildfire, and fire severity in these unfinished treatment units was very high (Safford, 2008; Safford et al., 2009; USFS, 2011). The Peterson Fire had different levels of treatment, and fire severity was lower in the treatment units that had been prescribe burned (Merriam, 2008); Prichard et al. (2010) found similar effects in a fire in Washington. Given the technical, logistical and environmental (principally air quality) issues associated with prescribed burning, some management units are defaulting to mastication. Mastication does not remove fuels from the site and although flame lengths and spread rates may be reduced (and fire control efforts aided), modeling and empirical evidence show that tree mortality can be very high in burning masticated fuels (Stephens and Moghaddas, 2005; Knapp et al., 2008; Safford, 2008).

Our data showed that tree mortality rates in treated areas that burned in wildfire were generally much lower than in neighboring untreated forest (mean of 22% vs. 73%). Thus, completed treatments notably reduced loss of tree carbon to wildfire, a result consistent with modeling and post-fire assessments of forest carbon (Hurteau and North, 2010; North and Hurteau, 2011). However, reduced tree carbon loss following wildfire must be viewed in the context of the carbon cost of biomass removal from treated areas before they encountered fire, as well as the ultimate use of that removed biomass (e.g., relatively long-term sequestration in building materials; biomass burning for energy, which supplants fossil fuels; pile burning on site, which releases carbon directly to the atmosphere, etc.). Mixed conifer and yellow pine forests in California supported tremendous amounts of fire before the arrival of Euroamerican settlers (Sugihara et al., 2006; Stephens et al., 2007; van de Water and Safford, 2011). Although most of that fire is generally understood to have been of lower severity, such a relationship with fire – especially when combined with projected increases in the inertia for fire as climates continue to warm – suggests that these forest types should not be focus areas for grand schemes to increase US carbon sequestration rates. Rather, in these forest types it makes sense to focus on management practices that restore fire- and drought-resilient forest structures that are more likely to *retain* tree carbon through recurrent fire (and other disturbance) cycles (Hurteau and North, 2009). Such practices, which focus on the recruitment and retention of large, fire-tolerant trees, include forest thinning of smaller individuals of more fire-sensitive species (with the removed biomass used wisely), prescribed burning, and an expansion of the use of naturally-ignited wildfire.

There were strong differences in survivorship among the tree species we sampled. Nearly nine of ten individuals of yellow pine (*P. jeffreyi* and *P. ponderosa*) survived fire in treated forest stands, and more than six in ten survived overall; *Quercus kelloggii* and *Pseudotsuga menziesii* also showed relatively high survival in treated stands. The first three were dominant taxa in Sierra Nevada yellow pine and mixed conifer forests before Euroamerican settlement, but in many places they have been supplanted by less fire tolerant, more shade tolerant species like *Abies concolor*. Dense stands of *A. concolor* are common in areas that have lacked fire for many decades. Fire-caused mortality in such fuel-rich forests can be very high, including among fire tolerant species that are embedded in the forest matrix.

Although our treatments spanned a range of one to nine years old when they were burned, we did not find an effect of treatment age on any of our measures of fire severity or tree survival. This is not a surprising result, since effective treatment lifespan should scale with the historical fire return intervals for the forest type in question. Collins et al. (2009) found that wildland fire use (WFU) fires in Yosemite National Park did not reburn previously burned areas until at least nine years had passed. They argued that this "self-limiting" behavior of fires was due to the period of time required to restore sufficient surface fuels to carry fire; after nine years, the probability of reburn was driven by fire weather conditions. Other authors have also estimated that treatment lifespans might be expected to range from 10–15 years in fire frequent forest systems like those we sampled (e.g., Graham et al., 2004; Agee and Skinner, 2005; Evans et al., 2011). In the future, we hope to expand our sample of fires to some that burned 10–20 years after treatment, so as to allow an empirical test of these rules-of-thumb.

In addition to fuel loading, fire weather and fuel moisture can be important to the effectiveness of forest thinning treatments. However, the relative importance of these factors varies along a number of environmental and ecological gradients. In their natural state (i.e., with recurrent, low severity fire), dry mixed conifer and yellow pine forests support classic "fuels limited" fire regimes (Agee, 1993; Schoennagel et al., 2004), where fire-season conditions are nearly always ripe for burning assuming there is enough fuel to carry fire. After nearly a century of fire exclusion and other management practices, many of these forests have filled with shade-tolerant tree species (e.g., *A. concolor*) and surface and ladder fuels to the point that their relationship with fire has become markedly more "climate limited" (Agee, 1993; Schoennagel et al., 2004; Miller et al., 2009), where sufficient fuel to carry fire is always at hand, and the occurrence of fire depends more on whether climate or weather is suitable for ignition and fire spread. Largely because they burn less often and thus accumulate fuel, climate-limited fire regimes burn at higher severities on average than fuels-limited fire regimes, and thus forests which have converted from the latter to the former have crossed a significant ecological threshold. Where they are properly implemented, we believe that completed forest fuel treatments in mixed conifer and yellow pine forests can reasonably replicate patterns of presettlement fire behavior (among other ecological patterns). At the simplest level, our study clearly shows that fire severity and tree mortality are generally much lower in fuel treatments. We also found that fuel moisture (a climate-driven variable) was a better predictor of tree survival in untreated forest stands (Fig. 6), while fuel loading (the amount of fuel) better predicted survival in treated stands. Taken together, these two patterns support the theoretical framework described above, and suggest that in places like many of our study sites, forest restoration strategies will need to focus on strategic reduction of fuels before frequent fires can be safely and beneficially reintroduced to the landscape.

Two of the fires we analyzed (Harding and Milford) did not show any statistically significant effects of fuel treatment on tree survivorship or fire severity. Tree survival in treated burned stands at these two sites was similar to the other fires (although measured survivorship in the Harding Fire was <60%), but the two fires showed lower mortality in untreated stands than any of the other 10 fires we assessed (Fig. 4). Tree scorch and torch data from the Milford Fire tell a similar story (Fig. 3), i.e., treated results are similar to other fires, but untreated fire severity is lower. Both Harding and Milford Fires occurred in relatively cold and dry areas at or east of the Sierra Nevada crest (Fig. 1, Table 1) that are dominated by Jeffrey pine and white fir. The two fires had among the lowest untreated-forest fuel loadings of any fires in our sample, and both fires occurred under relatively benign fire weather conditions (low ERCs, and relatively high fuel moistures; Tables 2a and 2b). The Harding Fire occurred under very moderate winds, and the Milford Fire burned in April when snow still covered some of the site. The difference in overall tree survivorship between the two fires may be due to: (1) differences in season of burning and tree susceptibility to injury; (2) time since fire – 5 years elapsed between the Harding Fire and our sampling, giving more time for delayed tree mortality; and (3) the fact that the Harding Fire was followed by a number of relatively dry years, which would presumably reduce tree survivorship.

Not all trees survive fire in treated forest, but this has potential ecological benefits. Lack of fire in some western forests may have led to a lack of fire-created structures like snags and down logs that provide important wildlife habitat (Brown et al., 2003; see Skinner (2002) and Lawler et al. (2011) for discussions of this effect in California frequent-fire forests). Estimates of presettlement fire severity and contemporary reference sites suggest that c. 5–15% of fire area typically burn at high severity (where >75% of the canopy trees are killed by fire) under "natural" fuel and fire conditions in mixed conifer and yellow pine forests (Stephens et al., 2007; Collins et al., 2009). When all mortality is taken into account however, >30% of (mostly small to medium, understory) trees may be expected to die even in a mostly low severity fire. This militates for accepting, and even planning for some level of mortality of canopy trees both in prescribed burns and wildfires that encounter fuel treatments. Based on the literature, an informal analysis of satellite-derived fire severity measures of the fires we sampled for this study, and an ongoing assessment of tree mortality patterns in contemporary frequent-fire reference ecosystems, we suggest that fire and resource managers should expect (and, indeed, plan for) 5–15% mortality of canopy trees in forest fuel treatments impacted by wildfire, as well as in prescribed fires conducted in similar sites; burning under severe conditions will obviously result in more extensive canopy tree mortality. Of course, even higher levels of canopy tree mortality should be expected in wildland fire use or prescribed fire in untreated stands.

Overall, our results support the central conclusions we made in an earlier paper that treated the Angora Fire on its own (Safford et al., 2009). These include the general efficacy of completed fuel treatments in reducing fire severity and tree mortality and the reduced loss of biomass/carbon from fire in treated areas, but there were other commonalities as well. For example, we found in the current study that steeper slopes had a stronger effect on fire severity in treated than in untreated forests. This is an extension of the effect we reported on in the Angora Fire, and is due to the influence of heavy surface and ladder fuels, which overwhelm the effects of underlying topography in driving fire behavior (Safford et al., 2009). Once the forest is thinned, slope becomes an important driver of fire behavior again. The current expanded study also corroborates the general patterns we saw in the Angora Fire vis-à-vis the linear distance required to reduce a canopy fire to a surface fire. In the fires we assessed, somewhere between 40–70 m was usually sufficient to realize such an effect. As argued in Safford et al. (2009), considering fire spread rates under extreme conditions (up to 3 km/h), fire fighter response times (>10 min, even in urban forests), and other complicating factors, 400–500 m (c. 1/4 mile) is probably a justifiable minimum width for fuel treatments expected to significantly slow or stop wildfire. Of course, this calculation ignores ember production. Recent work in Australian eucalypt forests determined that a minimum treatment width of 2000 m (and relatively severe levels of fuel reduction) was required to reduce the probability of home loss to "acceptable" levels (R. Bradstock, pers. comm.). Obviously, the intensiveness and extensiveness of strategic forest thinning will depend to a great degree on the ultimate objective(s) of treatment.

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