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Pre-wildfire fuel reduction treatments result in more resilient forest structure a decade after wildfire

Camille Stevens-Rumann[A,D], *Kristen Shive*[B], *Peter Fulé*[B] *and Carolyn H. Sieg*[C]

[A]College of Natural Resources, University of Idaho, PO Box 441133, Moscow, ID 83844-1133, USA. Email: csrumann@uidaho.edu
[B]School of Forestry, Northern Arizona University, PO Box 15018, Flagstaff, AZ 86011, USA. Email: kls448@nau.edu
[C]USDA Forest Service Rocky Mountain Research Station, 2500 Pine Knoll Drive, Flagstaff, AZ 86001, USA. Email: csieg@fs.fed.us
[D]Corresponding author. Email: csrumann@uidaho.edu

Abstract. Increasing size and severity of wildfires have led to an interest in the effectiveness of forest fuels treatments on reducing fire severity and post-wildfire fuels. Our objective was to contrast stand structure and surface fuel loadings on treated and untreated sites within the 2002 Rodeo–Chediski Fire area. Data from 140 plots on seven paired treated–untreated sites indicated that pre-wildfire treatments reduced fire severity compared with untreated sites. In 2011, coarse woody debris loading (woody material >7.62 cm in diameter) was 257% higher and fine woody debris (woody material <7.62 cm) was 152% higher on untreated sites than on treated sites. Yet, in spite of higher levels of coarse woody debris on untreated sites, loadings did not exceed recommended ranges based on published literature and many treated sites fell below recommendations. By 2011, basal area and stand density on treated sites and stand density on untreated sites met management guidelines for ponderosa pine forests, but untreated sites had basal areas well below recommendations. Snags declined over this period and only three plots had snags that met minimum size and density requirements for wildlife habitat by 2011. The effects of pre-wildfire treatments are long-lasting and contribute to changes in both overstorey and understorey fuel complexes.

Additional keywords: Arizona, coarse woody debris, ponderosa pine, Rodeo–Chediski Fire, snags, stand recovery.

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Introduction

There is widespread interest in the effectiveness of fuels reduction treatments in mitigating fire hazard, especially in the face of increasing size and severity of wildfires in recent decades (Westerling *et al.* 2006; Littell *et al.* 2009). Numerous studies have examined the effectiveness of various fuels treatments, both through modelling and field sampling. Most research has found that subsequent wildfire severity and crown fire susceptibility decreased consistently in forests that were thinned and burned to reduce tree density and stand basal area (Deeming 1990; van Wagtendonk 1996; Stephens 1998; Fulé *et al.* 2001, 2012; Brose and Wade 2002; Omi and Martinson 2002; Pollet and Omi 2002; Korb *et al.* 2012; Prichard and Kennedy 2012; Safford *et al.* 2012). Yet such treatments may not always be effective, especially in cases where thinning occurred without the removal or burning of slash (Raymond and Peterson 2005; Stephens and Moghaddas 2005; Prichard and Kennedy 2012). Additionally, in terms of wildlife values and soil protection, there is some trade-off to thinning, illustrated primarily by the decline in snag density and sometimes persistent surface fuel loadings below desired ranges (Raymond and Peterson 2005).

Immediate post-fire effects in treated *v.* untreated areas have been widely studied but few studies have analysed the longer-term effects of fuels treatments on stand dynamics and fuel loading. Most studies have focussed on burned sites shortly after the wildfire and used modelling projections to predict recovery trajectories on longer time scales (Pollet and Omi 2002; Finney *et al.* 2007; McIver and Ottmar 2007; Strom and Fulé 2007). Some studies have illustrated the longer-term changes in pine forests after wildfire through chronosequence studies and modelling (Foxx 1996; Gracia *et al.* 2002; Greene *et al.* 2004; Passovoy and Fulé 2006; Roccaforte *et al.* 2012; Stevens-Rumann *et al.* 2012) and some have shown that alternative ecotypes such as grasslands or shrublands may persist in severely burned areas (Barton 2002; Savage and Mast 2005). However, few studies have supplied empirical evidence for how fuels change over time on a single fire, particularly in relation to the effect of pre-wildfire treatment on post-wildfire fuels and stand structure over time. Understanding the effect of thinning and other fuels reduction treatments on post-wildfire recovery will be critical given recent trends of larger and more severe wildfires (Westerling *et al.* 2006; Littell *et al.* 2009). Thus, in the present study, we examined the effectiveness of fuel treatments longer term in reducing fuel loadings and thus preserving forest structures more resilient to wildfires.

The Rodeo–Chediski Fire, which burned 189 000 ha in 2002 and was at the time the largest wildfire in south-western US history, has been included in multiple studies on fuels and fire hazards. Cram and Baker (2003) concluded that pre-wildfire treatments as much as 20 years previously successfully decreased crown fire hazard. Finney *et al.* (2005) and Roccaforte *et al.* (2012) also included the Rodeo–Chediski Fire in their studies and discussed the dynamics of woody debris accumulation. Roccaforte *et al.* (2012) studied untreated areas of the Rodeo–Chediski Fire and found woody accumulation similar to loadings on other fires throughout Arizona. Finney *et al.* (2005) compared pre-wildfire treatments and found that the size of pre-fire treatments affected the longevity and effectiveness of reducing fire behaviour. In 2011, we remeasured seven paired treated–untreated sites installed in 2004 by Strom and Fulé (2007) to assess the differences between short- and longer-term fuel responses. Strom and Fulé (2007) found that untreated stands burned more severely, had lower tree survival and overall less pine regeneration than treated stands. For the present subsequent study, we asked: (1) how did fuel complexes differ between pre-fire-treated and -untreated areas post-wildfire (2004) and (2) how did fuel complexes (stand structure and surface fuels) change through time (2011)? We then used these results to ask: (3) how do permanently marked and tracked datasets compare with chronosequence studies and (4) how do these respective post-fire fuel complexes meet forest structure and wildlife habitat goals 9 years after wildfire?

Methods

Study area

The Rodeo–Chediski Fire burned along the Mogollon Rim in east-central Arizona. Climate averages for 1950–2012 from the Heber Ranger Station within the burn area were: July maximum temperature 29.2°C, January minimum temperature −8.7°C, and an annual precipitation of 43.9 cm (Western Regional Climate Center, http://www.wrcc.dri.edu, accessed 25 May 2012; Fig. 1). Elevation of the study sites ranged from 1990 to 2138 m. The soil type varies from clay substrates to sandy loams, depending on the parent material; alluvial gravels are present in drainages, and the Mogollon Rim itself has a limestone bed. Forests were dominated by ponderosa pine (*Pinus ponderosa*) with Gambel oak (*Quercus gambelii*) and New Mexico locust (*Robinia neomexicana*).

Study sites

After the wildfire, Apache–Sitgreaves National Forest managers identified 14 sites where non-commercial thinning for fuel reduction and slash disposal through pile burning were conducted between 1990 and 1999, of which seven were randomly selected for measurement. Untreated sites were chosen adjacent to the treated sites. The untreated sites had similar topography and pre-thinning forest structure, with no road or other firebreak in between to control for other causes of variable fire behaviour (L. Wadleigh and C. Hoffman, pers. comm., 2003, 2006). In 2004, Strom and Fulé (2007) established 10 plots in systematic grids within each of the seven paired study sites, for a total of 140 plots. Plots were permanently marked with a tagged iron stake sunk to ground level at each plot centre in 2004. We were able to relocate and measure 135 of the original 140 plots between May and August of 2011.

Measurements

Downed woody debris, and litter and duff depths were measured along a 15.2-m planar transect within each plot, using methods established in Brown (1974). Fine woody debris (FWD) was classified by size class (0–0.64, 0.65–2.54, 2.55–7.62 cm), which corresponds to 1-, 10- and 100-h moisture time-lag classes (Fosberg 1970). Coarse woody debris (CWD, or 1000-h fuels), were inventoried as logs >7.62 cm and categorised by sound and rotten classes (Fosberg 1970).

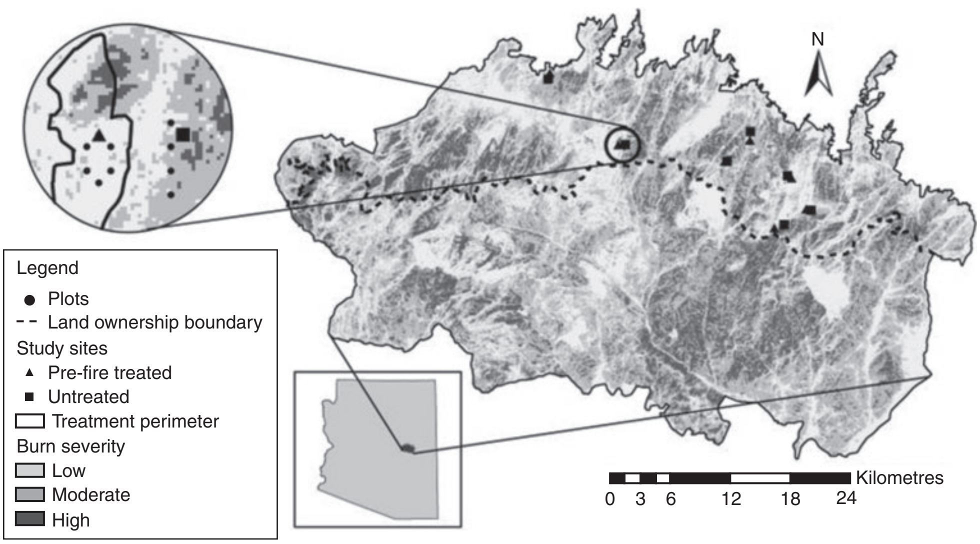

Fig. 1. Map of the Rodeo–Chediski Fire in east-central Arizona. Map taken from Shive *et al.* (2013*b*).

We measured overstorey trees on a variable-radius plot from the centre of each permanently marked plot using a prism with a basal area factor of 2.2 $m^2\,ha^{-1}\,tree^{-1}$. Tree measurements included species, condition and diameter at breast height (DBH). Tree condition classes followed Thomas' (1979) description: live, declining and four stages of snags (recent snag, loose-bark snag, clean snag and snag broken above breast height). We used a hemispherical fisheye lens on a digital camera (FC-E8 Fisheye Converter Lens and Nikon CoolPix E4300, Nikon, Melville, NY, USA) to photograph canopy cover and analysed these photographs using the *Gap Light Analyzer* (Frazer *et al.* 1999) to quantify percentage canopy openness. In addition, we rephotographed each plot overview horizontally to record a series of plot images through time.

Data analysis

In all analyses, sites were the experimental units, resulting in a total sample size of $n = 14$. Each plot, 135 in total, was treated as a subsample and all analyses were carried out as a paired design. We tested differences between treated and untreated sites in stand structure characteristics (canopy cover, basal area and tree density) and surface fuels (forest floor depth (sum of duff and litter depths), 1-, 10-, 100-, 1000-h sound and rotten). All residuals met the assumptions of parametric statistics, based on visual inspection of q-q plots and histograms. We used repeated-measures analysis of variance (ANOVA) with site as a blocking factor. Our model tested the main effects of treatment and year, plus year-by-treatment interaction. We also used a one-way ANOVA to test the treated *v.* untreated effect within each year separately and year effect on each treatment separately when we observed a treatment-by-year interaction ($\alpha = 0.05$). Additionally, owing to contract tree removal (G. Richardson, US Forest Service, pers. comm., 15 March 2012) that occurred on two of the seven untreated sites since 2004, we performed all individual-year analysis from 2011 without these sites to determine if there were significant differences as a result of this activity. Excluding these two sites did not change the significance of any variable; therefore, we chose to include these sites in our analyses to increase our degrees of freedom. We used *JMP* (SAS Institute Inc. 2007) for all these analyses. The *F*- and *P*-values shown in Figs 1–2 are from the repeated-measures analysis, whereas the *F*- and *P*-values expressed in the text are from the one-way ANOVAs.

Results

Surface fuels

All categories of surface fuels had significantly higher loadings on both treated and untreated sites in 2011, compared with 2004; however, forest-floor depth did not vary significantly ($P > 0.05$) between 2004 and 2011 in either treated or untreated sites. The increases in fuel loads over time were significantly greater on untreated sites than treated sites, with differences between treated and untreated sites becoming more pronounced by 2011, in all size classes except 10-h fuels. There was significantly more fuel on both untreated sites and treated sites in several woody fuel categories in 2011 than 2004, indicating a time-since-fire effect. We detected a significant year-by-treatment interaction in the repeated-measures analysis for 1- and 1000-h sound fuels; thus, we analysed data from the 2 years and two treatments separately. In all other surface fuel categories (10-, 100- and 1000-h rotten), there was not a significant treatment effect or an interaction effect. In 2004, mean loadings in all time-lag classes were slightly, but not significantly, higher on treated sites than on untreated sites. However, in 2011, mean fuel loadings in all size categories were higher on untreated sites with significant differences detected in 1-h fuels ($P = 0.0016$) and 1000-h sound fuels ($P = 0.0084$; Fig. 2). On untreated sites, loadings in all three of these fuel categories increased significantly between 2004 and 2011 ($P < 0.0008$), but there were no significant differences on treated sites across this time period ($P > 0.1$).

Forest floor depth did not vary significantly between treated and untreated sites in either year of sampling. Treated stands had slightly higher means, but the difference was not significant. There was a significant increase in depths between years from 1.4 ± 0.4 to 1.7 ± 0.7 cm on treated sites and from 0.9 ± 0.4 to 1.6 ± 1.2 cm on untreated sites.

Stand structure

The analysis of canopy cover revealed a significant year-by-treatment interaction due to the slight increase in percentage cover in treated areas and a decrease on untreated areas in 2011 compared with 2004. Thus canopy cover was significantly higher on treated sites than on untreated sites in the individual-year analysis of both 2004 ($P = 0.0044$) and 2011 ($P = 0.0002$), with larger mean differences in 2011. Between 2004 and 2011, canopy cover significantly declined ($P = 0.0087$) on untreated sites, but significantly increased on treated sites ($P = 0.013$; Fig. 3*a*).

Live basal area was relatively unchanged between 2004 and 2011, and ranged from 1.1 to 6 $m^2\,ha^{-1}$ on untreated sites and from 3 to 22 $m^2\,ha^{-1}$ on treated sites. These basal areas were significantly higher on treated sites than on untreated sites over both years. There was no significant year or year-by-treatment interaction between 2004 and 2011 (Fig. 3*b*). The differences between untreated and treated sites were not as pronounced in terms of live stand density, with no significant differences between treatments. The mean stand density ranged from 47 to 621 trees ha^{-1} on treated sites and 40 to 2166 trees ha^{-1} on untreated sites. As with basal area, there was not a significant year effect or a year-by-treatment interaction for live tree density (Fig. 3*c*).

There was a significant year-by-treatment interaction for snag basal area. Snag basal area was significantly higher on untreated sites than on treated sites in 2004 ($P = 0.0067$), but not by 2011 ($P = 0.078$). The mean snag basal area on untreated sites declined significantly from 2004 to 2011 ($P = 0.0002$); however, on treated sites the decline in snag basal area was not significant ($P = 0.071$; Fig. 3*d*). Snag density did not have a significant year-by-treatment interaction and was higher on untreated plots in the repeated-measures analysis. The year effect was also significant, with snag density significantly declining on treated and untreated sites from 2004 to 2011. This low observed snag density in 2011, on both treated and untreated sites, illustrated that few snags remained standing by this time (Fig. 3*e*). We also compared snag stages in 2004 with those recorded in 2011 and found that most snags were 'recent snags' in 2004, but by 2011 most were 'broken snags'. There were more snags observed in every category on untreated sites, in both years (Fig. 4*a*, *b*). Finally, in the analysis of snags only >40-cm

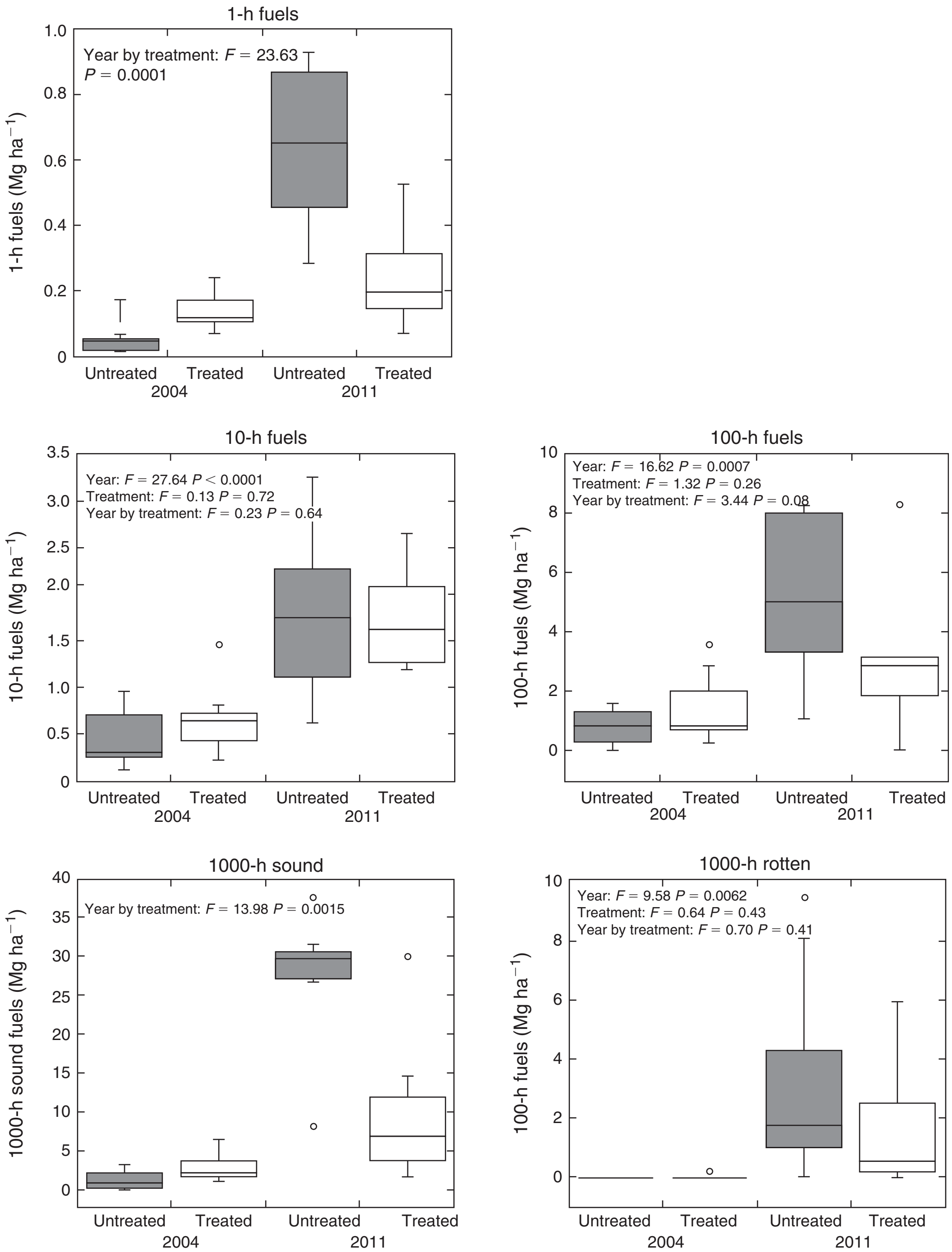

Fig. 2. Surface fuel characteristics: grey box plots indicate untreated sites; clear box plots indicate treated sites. Clear circles identify outliers and T-bars indicate the 75th percentile. The 2004 data are shown on the left of each graph and 2011 data are on the right. *F*- and *P*-values are from the repeated-measures ANOVA, showing the year, treatment and year-by-treatment interaction values.

DBH, we found mean snag densities of 5.82 snags ha^{-1} on untreated sites and 2.41 snags ha^{-1} on treated sites (Fig. 4*c*).

Discussion

Pre-wildfire treatments reduced original fire severity and a decade later continue to have significant, persistent effects on post-wildfire fuel complexes when compared with untreated sites. Tree mortality was high on untreated sites, which resulted in CWD loads averaging 257% higher than treated sites. However, these higher surface fuel loadings may not pose a major management or containment threat because they do not exceed recommended loadings (Brown *et al.* 2003).

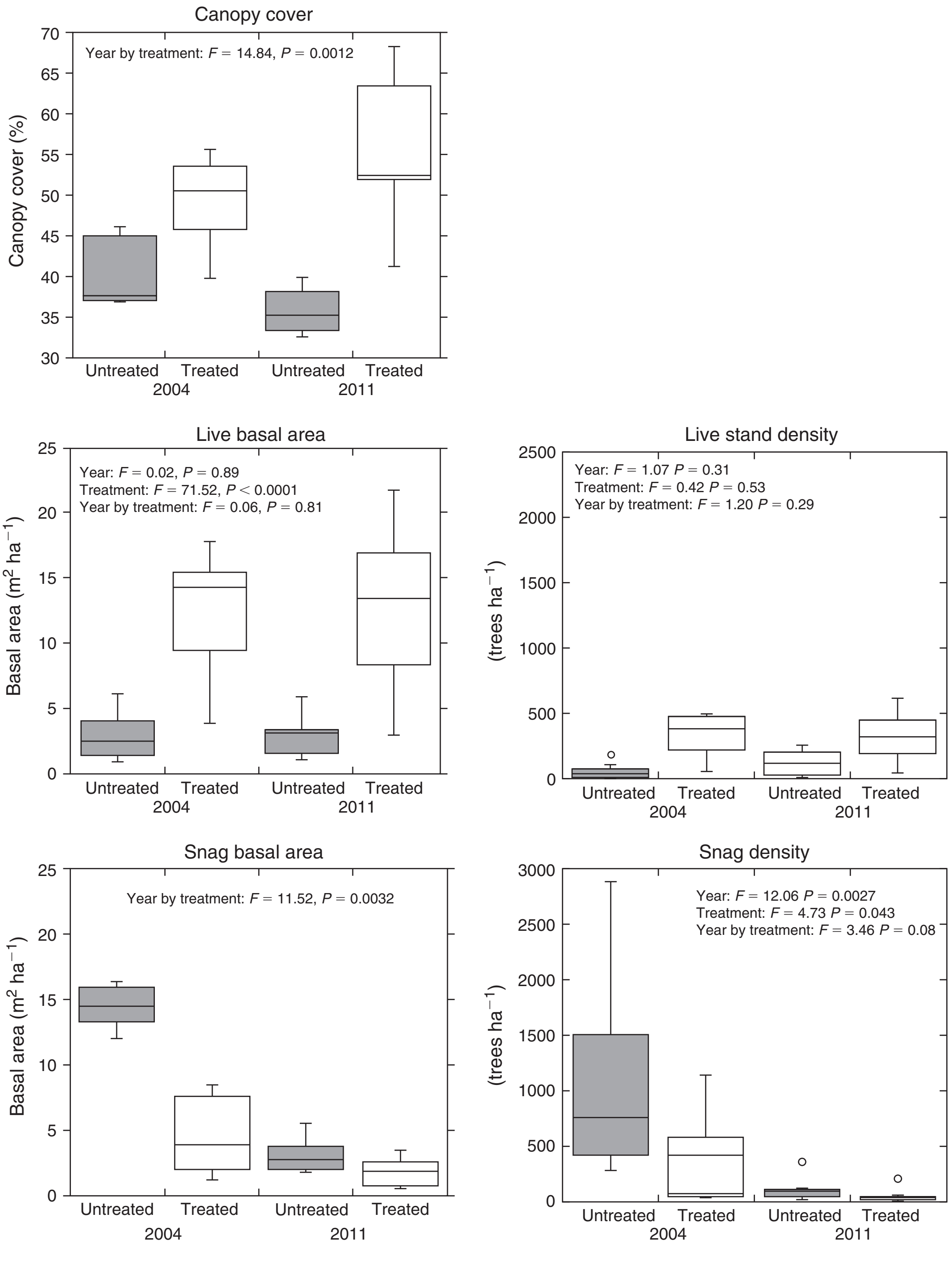

Fig. 3. Canopy fuel characteristics: canopy cover, live basal area and tree density, snag basal area and tree density: grey box plots indicate untreated sites; treated sites are represented by clear box plots. Clear circles identify outliers and T-bars indicate the 75th percentile. The mid-line indicates the mean. The 2004 data are shown on the left of each graph and 2011 are on the right. *F*- and *P*-values are from the repeated-measures ANOVA, for year and treatment main effects and year-by-treatment interaction.

Changes in fuel complexes

To address our first and second questions of how fuel complexes differed between pre-fire-treated and -untreated areas following a wildfire and how these complexes changed through time, we contrasted surface fuel loadings and stand structure characteristics. Higher surface fuel loads on both treated and untreated sites in 2011 compared with 2004 are a product of fire-killed trees falling over this time period. Pre-wildfire thinning did not have immediately discernible effects on post-fire surface fuels, but the difference became more pronounced over time as fire-killed trees fell, particularly in the untreated areas. Overall, the long-term effects of pre-wildfire treatment on surface fuels are substantial,

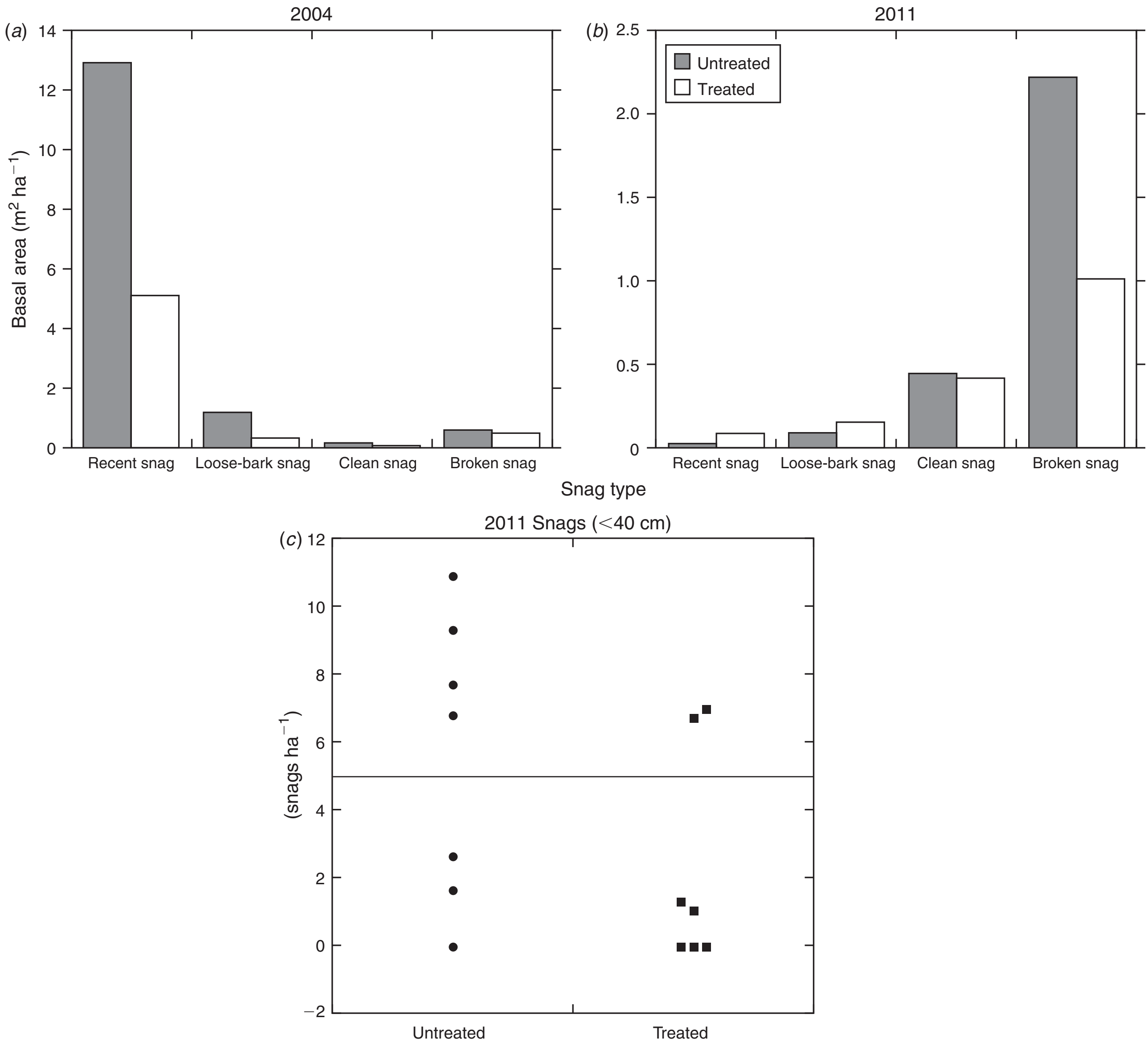

Fig. 4. Snag characteristics: snag type using the categories of Thomas (1979), their means separated by treatment in both years of sampling. Densities of snags >40-cm DBH (diameter at breast height). Each circle indicates the density on a single site. The line at 4.94 snag ha^{-1} indicates the target density of snags this size on the Apache–Sitgreaves National Forest (Apache–Sitgreaves National Forest Plan 1987, see http://www.fs.usda.gov/main/asnf/landmanagement/planning/ accessed 28 June 2012).

especially on the accumulation of 1000-h sound fuels. These findings were similar to other studies on pre-wildfire fuels treatments (Omi and Martinson 2002; Pollet and Omi 2002).

Stand structural attributes differed slightly over the study period. The observed increase in mean live basal area on treated sites was likely fostered post-wildfire by reduced overstorey cover and increased available resources (Fulé *et al.* 2005). However, in untreated sites, the mean live basal area decreased, indicating residual tree death occurred after the 2004 sampling. In similar fashion, canopy cover increased in treated areas from 2004 to 2011, whereas it decreased in untreated areas. Thus, treated sites are recovering and the canopy is filling in the space left open by fire-killed trees. However, untreated sites continue to have a smaller overstorey component with decreasing canopy cover and less basal area in 2011 than in 2004. This canopy recovery on treated sites and continued canopy decline on untreated sites is consistent with vegetation simulation model predictions made by Strom and Fulé (2007).

Snag basal area and density decline was similar to that observed by Chambers and Mast (2005) and Passovoy and Fulé (2006). Both of these studies showed that by 7 years post-wildfire, the majority of fire-created snags had fallen. In both years of sampling, untreated plots had more snags than treated plots. Snag decline lead to higher surface fuel loads from 2004 to 2011, with greater increases on untreated sites. Loadings in all fuel size classes significantly increased over the study period, illustrating the increase in fuel common in any forest without additional removal or fuel consumption from fire.

Table 1. Comparison among studies

Data were averaged across the fires within the specified time frame within each fire. 1–3 year data from Passovoy and Fulé (2006) are from 3 years after a wildfire and Roccaforte *et al.* (2012) present data from 1–2 years after wildfires. 7–11 year data from Passovoy and Fulé (2006) are from 8–9 years after wildfire and Roccaforte *et al.* (2012) have data from 7–11 years after fires. FWD, fine woody debris; CWD, coarse woody debris

	FWD ($Mg\,ha^{-1}$)	CWD ($Mg\,ha^{-1}$)	Forest floor depth (cm)	Snags (snags ha^{-1})	Basal area ($m^2\,ha^{-1}$)	Stand density (trees ha^{-1})
1–3 years						
Passovoy and Fulé (2006)	3.0	10.8	0.7	622.3	–	370.4
Roccaforte *et al.* (2012)	1	5	0.3	423	0.6	4.6
Rodeo–Chediski Fire	1.3	1.3	0.7	1108	3.0	61.3
7–11 years						
Passovoy and Fulé (2006)	7.8	36	1	124.6	–	81.5
Roccaforte *et al.* (2012)	7	60	0.8	220	2.7	1105
Rodeo–Chediski Fire	7.6	30.2	1.3	110	2.9	399.5
Change						
Passovoy and Fulé (2006)	4.8	25.2	0.3	−497.7	–	−288.9
Roccaforte *et al.* (2012)	6	55	0.5	−203	2.1	1100.4
Rodeo–Chediski Fire	6.3	28.9	0.6	−998	−0.1	338.8

Chronosequence comparison

To examine our third question of the effectiveness of chronosequence studies in predicting changes in fuel complexes over time, we compared our permanent plots with three studies within the South-west. Our primary comparison was with Passovoy and Fulé (2006), Roccaforte *et al.* (2012) and Stevens-Rumann *et al.* (2012) because these studies examined fires in northern Arizona and employed similar methods of fuels sampling. We compared fires within Passovoy and Fulé (2006) and Roccaforte *et al.* (2012) datasets that were 1–3 years and 7–12 years post-fire with our data from 2 and 9 years after the Rodeo–Chediski Fire. Stevens-Rumann *et al.* (2012) quantified fuel loading 10 years post-fire, so we used their data in our comparison of the later time period. None of these studies included areas that had pre-wildfire fuels reduction treatments so for this comparison, we only used untreated sites within our study (Table 1).

We found that immediately post-fire, FWD (<7.62 cm) was low in all studies, ranging from ~1 $Mg\,ha^{-1}$ (Roccaforte *et al.* 2012) to 3 $Mg\,ha^{-1}$ (Passovoy and Fulé 2006). Our data fell between these values at 1.3 $Mg\,ha^{-1}$. By 2011, FWD loadings had increased on our untreated sites to 7.6 $Mg\,ha^{-1}$, whereas Roccaforte *et al.* (2012) reported a mean loading of 7 $Mg\,ha^{-1}$ and the Passovoy and Fulé (2006) sites had mean loadings of 7.8 $Mg\,ha^{-1}$, with the highest FWD loadings on the Stevens-Rumann *et al.* (2012) sites at 10 $Mg\,ha^{-1}$. By this measure, it appears the increases in FWD were fairly consistent regardless of whether the study was a repeated-measures analysis or a chronosequence of fires.

CWD was lower on our sites in 2004 and 2011 compared with loadings in the other three studies. At both time periods, Passovoy and Fulé (2006) and Roccaforte *et al.* (2012) reported higher loadings based on their sites than observed on our sites. The fires studied by Passovoy and Fulé (2006) had the highest CWD loadings immediately post-fire in the three studies, but in the later time period, Roccaforte *et al.* (2012) and Stevens-Rumann *et al.* (2012) recorded loadings double those observed by both Passovoy and Fulé (2006) and us, with ~60 $Mg\,ha^{-1}$ CWD loads. The increase in CWD across the study period was the greatest in the case of Roccaforte *et al.* (2012), increasing by 55 $Mg\,ha^{-1}$, whereas our sites were similar to Passovoy and Fulé's (2006) sites with respective changes in fuel loads of 28.9 and 25.2 $Mg\,ha^{-1}$ in the same time period. This indicates that CWD loadings may be very fire-specific, depending on initial densities and the size of trees pre-fire. Unharvested ponderosa pine forests, like some of the study sites from Roccaforte *et al.* (2012), will create higher loadings following high tree mortality compared with high-density, small-diameter, second-growth forests studied in Passovoy and Fulé (2006) and our study (Table 1).

Snag densities also tended to be higher on the chronosequence sites that Passovoy and Fulé (2006) and Roccaforte *et al.* (2012) studied compared with our data at both time periods. Across the 7-year time period, snag density declined by 90% on our sites on the Rodeo–Chediski Fire, compared with an 80% decrease on fires studied by Passovoy and Fulé (2006) and a 50% decrease on the study sites of Roccaforte *et al.* (2012). Again, the differences in fall rates may be a result of the tree size, as Roccaforte *et al.* (2012) had sites with generally larger-diameter trees. This shows that fall rates across study sites are fairly similar, though highest in our study. These fall rates are also higher than those discussed by Chambers and Mast (2005), who showed a 41% decline in snags 7 years after a fire.

Trends in basal area and stand density varied in each study, but tended to be very low across all burned sites. Overall, basal area was very low on all study sites, with the only increase over the study period seen on the sites of Roccaforte *et al.* (2012). One to two years post-fire, mean basal area was 0.6 $m^2\,ha^{-1}$, whereas 7–11 years post-fire, mean basal area was 2.7 $m^2\,ha^{-1}$. In comparison, our untreated sites on the Rodeo–Chediski Fire had a mean basal area of 3.0 $m^2\,ha^{-1}$ in 2004, and 2.9 $m^2\,ha^{-1}$ in 2011, and thus were statistically unchanged over this time period and similar to basal areas from the sites of Stevens-Rumann *et al.* (2012), with basal areas of 2.5 $m^2\,ha^{-1}$ 10 years post-fire. Passovoy and Fulé (2006) did not provide basal areas, but presented high stand densities (370.4 trees ha^{-1}) on the fire that occurred 3 years before sampling, with significantly lower densities in older fires (81.5 trees ha^{-1}). We observed stand densities of only 61.3 trees ha^{-1} in 2004, which increased by 2011 to just under 400 trees ha^{-1}, which was consistent with increases observed by Roccaforte *et al.* (2012). Yet, in spite of the increases in stand densities in our study over the study period, the changes were not significant and were not substantial enough to change basal area at this time period. The changes in basal area observed by Roccaforte *et al.* (2012) are likely an artefact of sampling different fires or more prolific regeneration, rather than an observed trend of decreasing densities over the study period. Once again, these differences are due more to specific site conditions (even within a single fire), sporadic regeneration, pre-fire stand densities and fire severity and less to do with time since fire.

This comparison between chronosequences and repeated-measures studies illustrates the strengths and weaknesses of both types of studies, but also confirms that our study sites are similar to fires across ponderosa pine forests of the South-west. Establishing a baseline density and basal area for chronosequence studies is essential, as it will greatly affect the amount of tree survival, snag densities and ultimately differing surface fuel loadings over this time period, thus the accuracy of inferences from one burned area to another. We also advocate repeated-measures studies on existing plots whenever possible.

Meeting management goals

To address our fourth question, we assessed how post-fire fuel complexes met forest structure and wildlife habitat goals 9 years after wildfire. We compared post-wildfire fuel loads with acceptable ranges for management, as outlined in several scientific studies and National Forest guidelines.

To understand the implications of our CWD loadings, we compared our data with the 'optimum' CWD suggested by Brown *et al.* (2003). This 'optimum' range is a combination of the CWD needed for wildlife and understorey productivity, without presenting excessive fire hazard and soil heating potential (Brown *et al.* 2003), building on the management recommendations set forth by Graham *et al.* (1994). Unlike Roccaforte *et al.* (2012) and Fulé *et al.* (2002), we found that none of the sites exceeded the upper limit of 11.2–44.8 Mg ha^{-1}, and in fact several treated sites fell below the minimum of the target range. The lower CWD loadings (30.2 Mg ha^{-1} on untreated sites) may be due in part to post-fire firewood-cutting by the public as well as contract tree removal for fuel reduction that occurred on some sites since 2004. However, removing those plots and sites that we know experienced contract cutting only raised the mean CWD loading to 32.8 Mg ha^{-1}. Although we did not study the fuels spatial arrangement, overall CWD loadings do not appear to exceed recommended levels currently on treated or untreated sites. Because we do not expect CWD to continue to increase, owing to the minimal number of snags remaining, sites will not exceed the 'optimum' range in the future and may not be a source of concern for managers on either the treated or untreated sites. We did not measure live fuel biomass, but Shive *et al.* (2013*a*) reported total understorey percentage cover (regenerating trees, shrubs, forbs and graminoids). Total understorey cover averaged 18% in the untreated areas and 9% in the treated areas in 2011, suggesting live understorey species are unlikely to contribute substantially to fire spread.

To understand how the fire and thinning treatment affected the resulting live overstorey, we established thresholds for maximum basal area and tree density using similar methods to Stevens-Rumann *et al.* (2012). A basal area range of 9.2 and 18.4 m^2 ha^{-1} and tree density range of 140 and 250 trees ha^{-1} for ponderosa pine forests in the south-western US were based on historical norms for the region, as well as fuels reduction targets outlined by studies in this area of the US (Fulé *et al.* 1997, 2002). In 2011, treated sites fell within this basal area range, but untreated sites fell below the target range (Fig. 3*b*). This illustrates that the treatments increased tree survival post-fire, whereas untreated sites were well below the minimum target basal area owing to high tree mortality. However, stand densities on both treated and untreated sites were within the target range.

Snags provide ecosystem functions as habitat for cavity-nesting birds and other wildlife (Bull *et al.* 1997); thus, we examined the densities of larger-diameter snags because those are often considered of most value. We examined quantities of ponderosa pine snags >40-cm DBH, as this is the minimum snag size desired for many National Forests in the region (Kaibab National Forest Plan 1986, see http://www.fs.usda.gov/main/kaibab/landmanagement/planning/, accessed 25 May 2012; Apache–Sitgreaves National Forest Plan 1987, see http://www.fs.usda.gov/main/asnf/landmanagement/planning/ accessed 28 June 2012). The higher mean snag density on the untreated sites is unsurprising given that they also experienced a higher rate of tree mortality. The low mean snag density on treated sites may illustrate how pre-wildfire treatments can be a trade-off between conflicting resource-management objectives. Such treatments are beneficial in creating fire-resilient forests, yet they can be also detrimental, if thinning prescriptions result in fewer snags that are valued for wildlife (Stephens and Moghaddas 2005; McIver *et al.* 2013). The observed large snags were primarily ponderosa pine, with several Gambel oak and alligator juniper (*Juniperus deppeana*) and occurred on only a few plots. A total of 84% of the treated plots and 80% of the untreated plots lacked snags >40-cm DBH, meaning, overall, few large snags existed across sites. The average large-snag density met the desired density of ~5 snags ha^{-1} on untreated sites but fell below this density on treated sites (Apache–Sitgreaves National Forest Plan 1987; Fig. 4*c*).

Pre-wildfire thinning and pile-burning treatments decreased fuel loading and increased tree survival compared with untreated areas 9 years post-wildfire. These trends persisted and the differences between treated and untreated areas were more prevalent over time. Treatments such as these can ultimately change the stand trajectory after a large wildfire. Given that most forests of the south-western US are at high risk of severe crown fire (Fiedler *et al.* 2002; Schmidt *et al.* 2002), extensive fuel reduction treatments are needed to reduce the potential for high-severity fires. We have illustrated, as have many other studies across the western US, that pre-wildfire fuels treatments can result in reduced fire severity and long-term differences in stand recovery. Even in the face of extreme fire events that occur during severe fire weather conditions, like the Rodeo–Chediski Fire and more recent fires like the Wallow Fire of 2011 and the Whitewater–Baldy Fire of 2012, which are likely to increase owing to climate warming

(Williams *et al.* 2010), there is evidence that pre-wildfire fuels treatments help maintain forest structure and reduce future surface fuel loadings compared with untreated sites.

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