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Pre-wildfire fuel treatments affect long-term ponderosa pine forest dynamics

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Abstract. The 2002 Rodeo–Chediski fire, the largest wildfire in south-western USA history, burned over treated stands and adjacent untreated stands in the Apache–Sitgreaves National Forest, setting the stage for a natural experiment testing the effectiveness of fuel reduction treatments under conditions of extraordinary fire severity. In seven pairs of treated–untreated study sites measured 2 years after the fire, thinning was strongly associated with reduced burn severity. Treated areas had more live trees, greater survival, and reduced fire intensity as indicated by crown base height and bole char. Ponderosa pine regeneration was patchy but more dense in treated areas. We assessed decade- to century-long effects of the pre-wildfire fuel treatments using the Forest Vegetation Simulator (FVS). Differences between treated and untreated areas were projected to persist for several decades after the fire in terms of stand structure characteristics and for at least 100 years in terms of species composition, with ponderosa pine making up ~60% of basal area in treated areas but only 35% in untreated areas. Future ecosystem development may take the trajectory of recovery to a ponderosa pine/Gambel oak forest or of a shift to an alternative stable state such as an oak-dominated shrubfield, with untreated areas more apt to undergo a shift to a shrubfield state. Current management decisions about fuel treatments have multi-century legacies.

Additional keywords: Arizona, Forest Vegetation Simulator, fuel reduction, Rodeo–Chediski fire.

Introduction

Wildfires in frequent-fire-adapted ecosystems of the western USA are becoming larger and more severe in part due to changes in forest structure caused by fire suppression, selective logging of large trees, and livestock grazing occurring after extensive European–American settlement (Covington and Moore 1994; Swetnam *et al*. 1999). Stand treatments such as pre-commercial thinning, prescribed burning, and restoration to pre-settlement conditions have been projected to decrease the susceptibility of treated stands to crownfire due to their reduction of forest floor fuels, ladder fuels and canopy fuels, which can prevent fire from reaching the canopy and from transitioning from a passive crownfire (torching) to an active crownfire (Deeming 1990; van Wagtendonk 1996; Stephens 1998; Covington *et al*. 2001; Fulé *et al*. 2001; Brose and Wade 2002).

Several short-term empirical studies have examined adjacent treated and untreated stands after wildfire, reporting that areas that underwent fuel reduction treatments experienced lower burn severity (Agee *et al*. 2000; Martinson and Omi 2003). Some were non-systematic investigations, focussed on slash treatments rather than prescribed burning and thinning, or were based solely on remotely sensed data without field verification (Wagle and Eakle 1979; Omi and Kalabokidis 1991; Vihanek and Ottmar 1993). Weatherspoon and Skinner (1995) found that burn severity was greater in commercially harvested stands, but the stands they sampled had not been pre-commercially treated. Because conventional timber harvesting without pre-commercial thinning generally does not reduce ladder fuels, which contribute to crownfire susceptibility, it cannot be strictly considered a fuel reduction treatment (Agee and Skinner 2005).

Pollet and Omi (2002) and Omi and Martinson (2002) systematically examined the effect of fuel reduction treatments for eight wildfires, six of these in western ponderosa pine. Both studies found that burn severity and crown scorch were significantly lower in all treated areas. These studies can be used to make a more generalised statement about treatment effect on burn severity. Because the treatments for each wildfire differed, however, treatment effect cannot be clearly separated from the effect that the location and seasonality of the wildfire may have had on burn severity. Pollet and Omi (2002) questioned whether fuel treatments would have any effect under extreme fire conditions, as drought and high winds may play a more important role in fire behaviour than fuels. Cram and Baker (2003) also systematically investigated silvicultural treatment effect after four wildfires, including the Rodeo–Chediski burn area within the Apache–Sitgreaves National Forest. They found that treated areas up to 20 years old experienced lower burn severity, ground char, and fireline intensity as estimated from bole char.

There has been extensive empirical research on the long-term changes in pine-dominated forests after wildfire (Foxx 1996; Barton 2002; Gracia *et al*. 2002; Greene *et al*. 2004; Savage and Mast 2005), and several studies have also modelled forest dynamics after wildfire, though not all in pine forests (He *et al*. 2002; Retana *et al*. 2002; Chapin *et al*. 2003). Several of these long-term studies of forest recovery after wildfire have questioned whether forests that historically had a frequent fire

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Table 1. Tree and shrub species found on treated and untreated sites on the Apache–Sitgreaves National Forest

The columns '% frequency' indicate the percentage of plots on which the species was found

Common name	Scientific name	% Frequency	
		Overstorey	Regeneration
Tree species			
Ponderosa pine	*Pinus ponderosa*	98.57	6.43
Alligator juniper	*Juniperus deppeana*	35.00	31.43
Gambel oak	*Quercus gambelii*	33.57	65.00
Utah juniper	*Juniperus osteosperma*	1.43	0
Scrub (turbinella) oak	*Quercus turbinella*	0.71	3.57
Chihuahua pine	*Pinus leiophylla*	0.71	0
South-western white pine	*Pinus strobiformis*	0.71	0
Other oaks (unidentified)	*Quercus* spp.	0	6.43
New Mexico locust	*Robinia neomexicana*	0	3.57
Shrub species			
Fendler's ceanothus	*Ceanothus fendleri*	18.57	
Pinemat manzanita	*Arctostaphylos pungens*	12.86	
Pringle manzanita	*Arctostaphylos pringlei*	0.71	
Mountain mahogany	*Cercocarpus montanus*	0.71	

regime are resilient to crownfire, adding to the widespread concern about future development of arid forests such as those of the South-west as large, severe crownfires continue (Hessburg *et al*. 2005). Studies in Arizona (Barton 2002), Mexico (Fulé *et al*. 2000), and Spain (Retana *et al*. 2002) have indicated that intense fire in pine–oak forests may result in a shift to a more oak-dominated forest or a conversion to an alternative steady-state such as a shrubfield or grassland. These shifts in species dominance and conversions to what appears to be an alternative stable state have been documented after crownfire in several dense ponderosa pine forests by Savage and Mast (2005). Alternative stable states are self-perpetuating species assemblages distinct from the typical assemblage found in a given environment; they may be shorter in height (e.g. shrubs *v.* trees) and appear to be an earlier sere, though without evidence of near-term shift back toward the pre-crownfire forest. A shift to such an alternative state could have substantial consequences for ecosystem functions and potential land uses. For instance, some wildlife species such as tassel-eared squirrels are dependent upon ponderosa pine, and Gambel oak is much less economically valuable than the ponderosa pine it could replace as the dominant tree species (Barger and Ffolliott 1972; Blatner and Govett 1988; Linhart 1988; Petraitis and Latham 1999).

We selected the 189 000-ha Rodeo–Chediski fire of 2002, the largest severe wildfire in south-western history. We measured the effects of pre-fire fuel reduction treatments on seven paired treated–untreated sites, a relatively large and consistent dataset, sufficient to support reliable assessment of short- and long-term ecological response. We hypothesised that: (1) pre-wildfire thinning would lead to greater tree survival compared to that in untreated areas; (2) post-fire regeneration and shrubs, and thus potential forest development, would display higher density of sprouting species in untreated areas; (3) effects of forest structural and compositional differences would persist for at least several decades following the fire; and (4) untreated areas would be more likely to transition from ponderosa pine forest to an oak-dominated shrubfield.

Methods

Study area

The Rodeo–Chediski fire's 189 000-ha span the Mogollon Rim in east-central Arizona, including parts of the Apache–Sitgreaves National Forest and White Mountain Apache tribal lands. This present study focussed on the portion of the burn area in the Apache–Sitgreaves National Forest. Climate averages for 1971–2000 from the Heber Ranger Station on the north-western edge of the burn area (Western Regional Climate Center, http://www.wrcc.dri.edu, accessed 15 January 2005) were: July maximum temperature 29.2°C, January minimum temperature −7.8°C, annual precipitation 50.6 cm, and annual snowfall 99.3 cm (1950–2004 average). The study sites ranged in elevation from 1990 to 2138 m. The soil type varied from clay substrates to sandy loams, depending on the parent material; alluvial gravels were present in drainages, and the Mogollon Rim itself has a limestone bed. Forests were dominated by ponderosa pine with Gambel oak and New Mexico locust. Scientific names of all species are given in Table 1.

Study sites

We sampled seven pairs of treated–untreated stands on the Apache–Sitgreaves National Forest in May to August of 2004, 2 years after the Rodeo–Chediski fire (Fig. 1). To assess the relationship between pre-fire fuel reduction treatments and fire severity, shortly after the fire was controlled in 2002, Forest Service staff assembled information on all the areas within the burn perimeter that had been treated within 16 years before the fire. A total of 14 sites met the criteria of having maps of the treatment, adjacent untreated areas of similar topography, and no road or other potential firebreak between treated and untreated areas. Treated sites underwent non-commercial thinning for fuel reduction followed by slash disposal (pile and burn, except for one site where slash was lopped and scattered and one where slash was crushed) between 1990 and 1999. Residual forest stands were predominantly evenly spaced (average 3.8 m), small

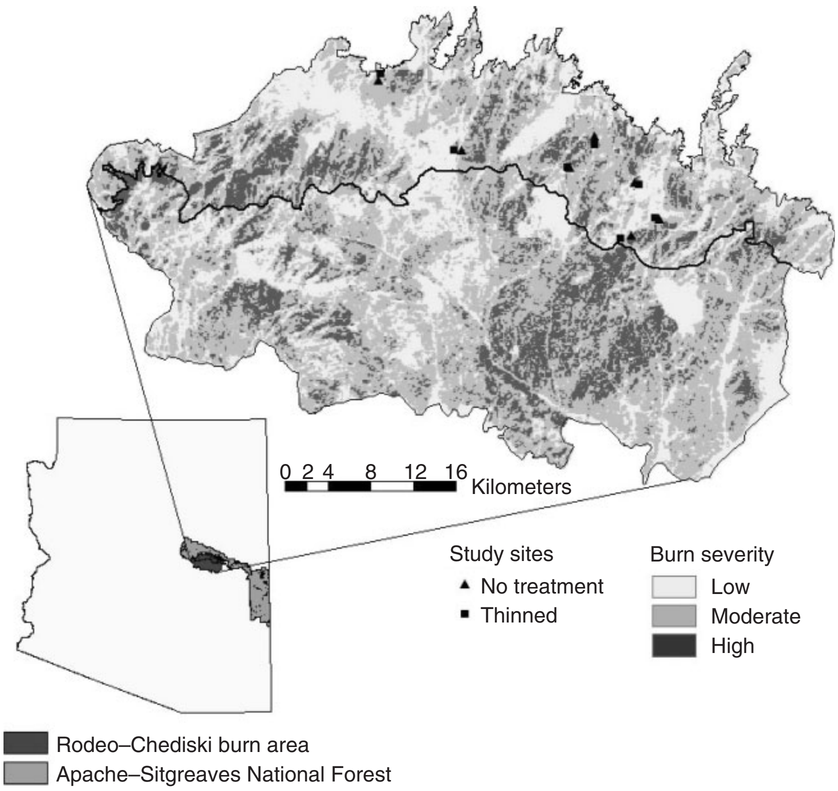

Fig. 1. Seven pairs of treated and untreated sites were sampled on the portion of the Rodeo–Chediski burn area within the Apache–Sitgreaves National Forest.

(modal diameter 20 cm) ponderosa pines. Seven treated sites and six paired untreated sites were sampled post-fire by the Forest Service (Fig. 1); the seven measured sites were selected randomly from the 14 potential sites. Untreated sites were selected from adjacent stands with no evident difference in topography, fuelbreak, or pre-thinning forest structure (USDA Forest Service 2002; L. Wadleigh and C. Hoffman, pers. comm. 2003, 2006). We established a seventh untreated site in 2004 near the pre-existing unpaired site. We located 10 plots in two systematic grids of five plots at each study site for a total of 140 plots. Each grid consisted of plots at the four corners of a 100×100 m square and the fifth plot in the centre. Plots were located using a Garmin Global Positioning System unit. Plots were permanently marked with a tagged iron stake sunk to ground level at each plot centre.

Measurements

We measured overstorey trees on a variable-radius plot delineated using a prism with a basal area factor of 2.2 m^2 ha^{-1} per tree. Tree measurements included species, condition, diameter at breast height (dbh), total height, canopy base height, bole char height (minimum and maximum), and dwarf mistletoe rating (0–6) using Hawksworth's (1977) scale. 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 did not have direct measurements of fire severity, but we inferred that bole char height was correlated with fire intensity and we used the proportion of tree mortality as a relative measure of fire severity. We did not attempt to estimate foliar scorch because field measurements were taken 2 years after the fire, so the majority of scorched needles had already fallen. We collected increment cores from a subsample of live trees (the live specimens of the first four trees on the plot; trees were numbered starting at north and proceeding clockwise around the plot) for age and growth increment data. Tree increment cores were surfaced and cross-dated (Stokes and Smiley 1968) or rings were counted for cores that could not be cross-dated, such as juniper cores. For cores that missed the pith, additional years to the centre were estimated with a pith locator (Applequist 1958). Ten-year diameter growth increments were measured (1994–2003, inclusive). We added living and dead trees together to estimate pre-fire forest density and basal area.

Tree regeneration (sprouts and seedlings below breast height) and shrubs were measured on a 0.00405-ha plot (3.6 m radius) with origin at plot centre. Tree regeneration and shrubs were tallied by species, condition (living or dead), and height class (0–40 cm, 41–80 cm, 81–137 cm, or exact height of shrubs if $>$137 cm). We measured forest floor fuels on a 15.2-m planar transect at a random azimuth from each plot centre, using Brown's (1974) method. We converted litter and duff depth to forest floor fuel loadings in Mg ha^{-1} using coefficients from Ffolliott *et al*. (1968).

Forest simulation modelling

Models that predict forest development fall into two general classes: statistical models, based on well-measured growth patterns, which can be quite accurate over short time periods; and, process models, which simulate physiological processes involved in growth and competition and thus can incorporate environmental change, but require much more complex initiation parameters. We used the Forest Vegetation Simulator (FVS), a statistical model of tree growth and yield (Dixon 2003). It is initialised with standard mensurational data, and outputs both stand-level and tree-level growth data by species. FVS has a locally developed south-western ponderosa pine model with high precision (Edminster *et al*. 1991). Other species, such as oaks, have been less well studied, so model outputs are likely to be more uncertain.

We also considered using FVSBGC, which combines FVS with STAND-BGC, a modified version of the FOREST-BGC biogeochemical process model (Running and Coughlan 1988; Running and Gower 1991). FVSBGC allows modelling of forest response to different climate and CO_2 scenarios (Milner *et al*. 2003). However, test runs indicated that the model requires further calibration before it can be used in south-western ponderosa pine.

The Central Rockies/south-western ponderosa pine variant of FVS was used to simulate stand development for each treated and untreated site for the next 100 years (2004–2104). We scaled the diameter increment model with measured 10-year growth increments and developed height–age relationships from our core data to reduce the site index from the default of 21.4 m per 100 years to 15.2 m per 100 years. Oak, juniper, and pine species were grouped and New Mexico locust was treated as a generic hardwood species for the purposes of the simulation.

We compared two different regeneration scenarios. The first, Regen-1, was simply the regeneration by species measured in 2004. The second scenario, Regen-2, also used the measured regeneration in 2004 but added a second regeneration event in 2024 based on average regeneration occurring 2–3 decades following crownfire in ponderosa pine forests as reported by Savage and Mast (2005). Ponderosa pine density was tripled and on sites where there were fewer than 50 stems ha^{-1} of the smallest height class, we introduced 150 stems ha^{-1}. This did not increase ponderosa pine regeneration density in 2024 to a level equivalent with the average reported by Mast and Savage (~950 stems ha^{-1}), but it fell within the range they reported (117–2864 stems ha^{-1}). It would have been necessary to increase our measured ponderosa pine regeneration by a factor of 28 to reach their average, and we did not believe such a large increase was justifiable. In Regen-2, juniper regeneration was decreased by a factor of 0.15, and oak regeneration was decreased by a factor of 0.65. Since the abundance of New Mexico locust on our study sites was similar to that reported by Savage and Mast (2005), we did not alter regeneration density for that species in Regen-2.

For each species and height class, expected survival percentages were estimated for both regeneration scenarios using measurements of regeneration before restoration treatments for two northern Arizona ponderosa pine forests (Fulé *et al*. 2001; Waltz *et al*. 2003). Ponderosa pine survival was 100% for all height classes, oak survival ranged from 9 to 45%, juniper survival ranged from 19 to 67%, and New Mexico locust survival ranged from 65 to 72%.

Data analysis

We compared forest structural data in treated and untreated areas using DISTLM (Anderson 2004), which performs a distribution-free, distance-based multifactor multivariate analysis of variance using permutation. We used the Bray–Curtis dissimilarity distance measure and carried out 999 permutations for each test. Since the method used by DISTLM calculates an exact *P*-value, tests are not adjusted for α inflation (Anderson and Robinson 2001). When testing overall regeneration and shrubs, we performed both an unstandardised test and a test standardised by species, in case the dominant species were skewing the results.

We used 95% confidence intervals to discern meaningful differences between modelling results for treated and untreated areas, and between the two regeneration scenarios.

Table 2. *P*-values and R^2 (proportion of variation explained) for comparison of treated *v.* untreated sites

Characteristic	*P*-value	R^2
Trees per ha (live)	0.001	0.2211
Trees per ha (live + dead)	0.966	0.0012
Survival (based on trees per ha)	0.001	0.2637
Basal area (live)	0.001	0.2622
Basal area (live + dead)	0.781	0.0023
Crown base and bole char height, live	0.005	0.0740
Bole char height, all	0.001	0.2325
Diameter distribution (live)	0.001	0.1760
Diameter distribution (live + dead)	0.856	0.0033
Snags	0.001	0.1758
Fuels, grouped by forest floor/fine/coarse	0.001	0.0493
Regeneration, all (unstandardised)	0.097	0.0132
Regeneration, all (standardised by species)	0.083	0.0131
Ponderosa pine	0.078	0.0196
Gambel oak	0.128	0.0129
Shrubs, all (unstandardised)	0.250	0.0089
Shrubs, all (standardised by species)	0.359	0.0070
Arctostaphylos pungens	0.043	0.0246

Results

Pre- and post-fire forest structure

Treated areas were strongly associated with lower tree mortality when compared to untreated areas. Pre-fire diameter distributions in treated and untreated areas (as estimated by summing living + fire-killed trees) were not significantly different according to the characteristics we tested, but post-fire conditions were very different (Tables 2 and 3). There were many more surviving trees in treated areas, and fire behaviour as indicated by bole char height was less extreme than in untreated areas.

While average tree density was substantially greater in untreated areas before the fire, there was no significant difference between treated and untreated areas. About half the trees in treated areas survived the fire, compared to 5% in untreated areas (Fig. 2). Post-fire density of live trees was significantly greater in treated areas (Table 2, Fig. 2). At least some trees survived on all of the study sites, but 66% of untreated plots and 11% of treated plots had no live trees. Basal area was not significantly different before the fire, but was significantly greater in treated areas afterwards (Table 2, Fig. 2).

Crown base height and bole char height for live trees and for all trees (living + dead) were significantly lower in treated areas, indicating less intense fire behaviour (Table 3). Bole char height for all trees was much higher than bole char for live trees, suggesting that most surviving trees were in pockets of lower than average fire intensity. The fire probably did not increase crown base height in treated areas, since crown base height was

Table 3. Overstorey, forest floor, and tree regeneration characteristics in treated *v.* untreated sites

Characteristic	Treated		Untreated	
	Mean	s.e.	Mean	s.e.
Crown base height (m)	6.6	0.3	8.6	0.7
Bole char on live trees (m)	2.0	0.3	4.0	0.7
Bole char on all trees (m)	2.9	0.4	9.8	0.7
Snag density, trees >30 cm dbh (trees ha^{-1})	11.4	2.9	39.5	5.1
Snag density, trees >50 cm dbh (trees ha^{-1})	1.9	0.5	6.5	1.0
Forest floor biomass (Mg ha^{-1})	10.1	0.7	6.5	0.7
Fine woody debris, <7.62 cm diam. (Mg ha^{-1})	2.2	0.4	1.3	0.3
Coarse woody debris, >7.62 cm diam. (Mg ha^{-1})	2.3	0.7	1.0	0.4
Juniper regeneration (stems ha^{-1})	670	300	1080	260
Gambel oak regeneration (stems ha^{-1})	1840	320	2080	400
Scrub (turbinella) oak regeneration (stems ha^{-1})	56	35	18	18
Other oak regeneration (stems ha^{-1})	21	16	56	28
New Mexico locust regeneration (stems ha^{-1})	250	180	510	350
Ponderosa pine regeneration (stems ha^{-1})[A]	53	22	14	11
<40 cm tall	35	15	4	4
40.1–80 cm tall	7	5	7	7
80.1–137 cm tall	11	8	4	4
Manzanita density (stems ha^{-1})	180	120	3860	1780
Fendler's ceanothus (stems ha^{-1})	680	340	490	200

[A]Total mean may not equal sum of means due to rounding.

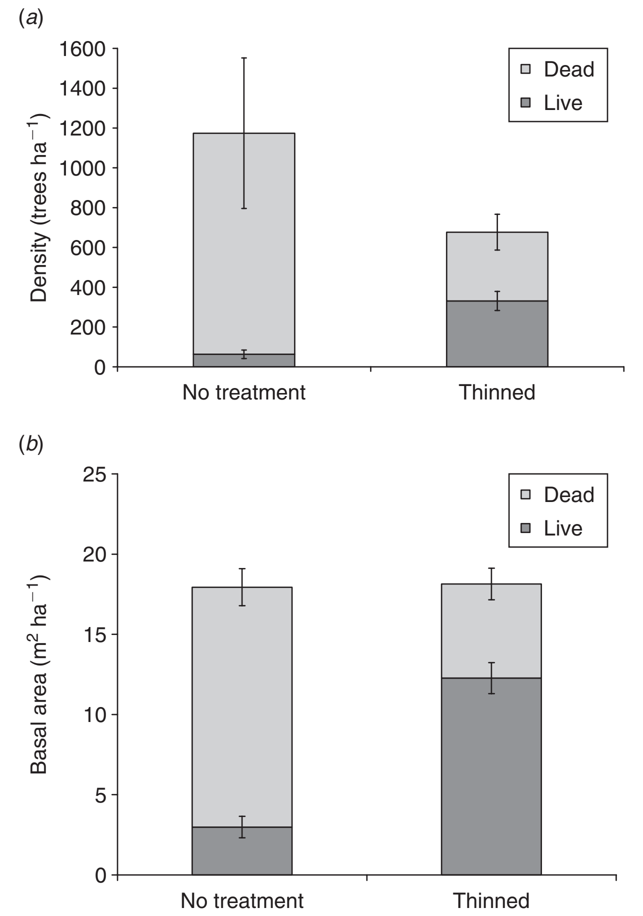

Fig. 2. (*a*) Density and (*b*) basal area of paired sites. Dead trees are presumed to have been alive before the fire.

substantially higher than bole char height for all trees. It appears to have done so in untreated areas, especially as crown base height there was higher than in treated areas.

There were far more small trees in untreated areas than treated areas before the fire, though pre-fire diameter distributions were not significantly different. The fire shifted the distribution in untreated areas strongly towards larger trees, but did not substantially change the shape of the distribution in treated areas (Fig. 3). About 95% of the surviving trees in both treated and untreated areas were less than 100 years old, though treated areas may have had slightly more old trees (data not shown). Many more standing snags were present in untreated areas of two common size classes used by many wildlife species (>30 cm dbh and >50 cm dbh) (Table 3).

Initial post-fire recovery

Initial post-fire recovery was relatively similar between treated and untreated areas. Only fuel loadings and manzanita density were significantly different. Fuel loading in terms of fine and coarse woody debris, as well as forest floor weight, were substantially greater in treated areas (Table 3).

Regeneration was dominated by sprouting species such as Gambel oak and alligator juniper, as well as by New Mexico locust (Table 3). Regeneration levels were slightly, but not significantly, more dense in untreated areas. Regeneration was present on every study site and 86% of the plots. Oak regeneration was present on all study sites and 68% of the plots.

On average, there was four times as much ponderosa pine regeneration in treated areas (Table 3), but there was no significant difference between treatments because of its patchiness. Ponderosa pine regeneration was found on only 6% of our plots; on the site level, eight of 14 sites (five untreated, three treated) had no ponderosa pine regeneration. In both treated and

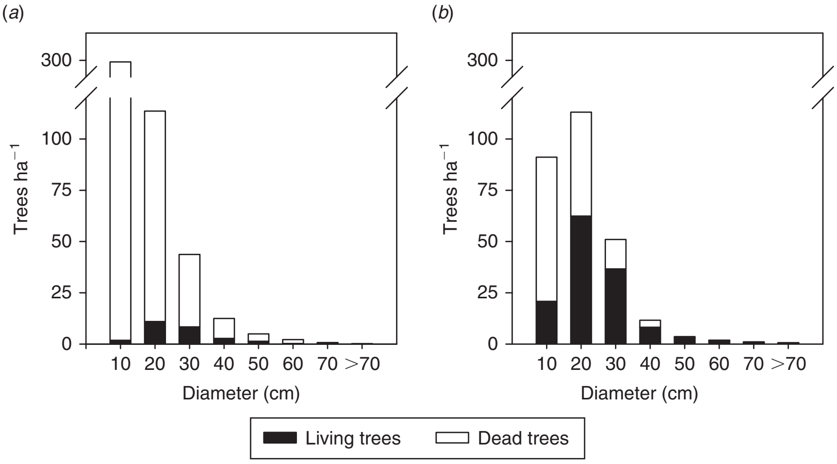

Fig. 3. The fire shifted the diameter distribution for ponderosa pine in favour of larger trees in untreated areas, but did not greatly change it in treated areas. (*a*) Unthinned diameter distribution, (*b*) thinned diameter distribution.

untreated areas, small amounts of ponderosa pine regeneration in the larger height classes appeared to have survived the fire. Live ponderosa pine trees were present on all sites, but 41% of the plots (67% untreated, 14% treated) had no surviving ponderosa pines. Sixty-four percent of untreated plots and 11% of treated plots had no surviving pines and no ponderosa pine regeneration.

Pinemat manzanita and Fendler's ceanothus (Table 3) were virtually the only shrubs present. While, overall, shrubs were not significantly different on treated *v.* untreated areas, manzanita was 20 times more abundant in untreated areas, and so was significantly different when considered separately.

Modelling

Differences between treated and untreated areas will likely persist for at least the next several decades in terms of overall forest structure characteristics (density, basal area, and quadratic mean diameter) as indicated by our modelling of future forest development using FVS. However, for the most part these differences were characterised by wide 95% confidence intervals (Fig. 4). Differences in relative species abundance and dominance endured or increased over the entire 100-year simulation period (Figs 4 and 5).

Treated areas initially had more trees, but as untreated areas had more regeneration, they quickly became denser; this difference slowly declined over the course of the simulation. All treatment and regeneration combinations led to some self-thinning, but Regen-2 (scheduling measured regeneration in 2004 and adjusted regeneration in 2024) in untreated areas led to an especially high pulse of density and a correspondingly steep decline. After 100 years, treated and untreated areas were nearly identical, but density under Regen-2 remained greater than that under Regen-1 (scheduling measured regeneration only in 2004). Confidence intervals overlapped for the entire simulation, indicating that the treatments were only meaningfully different at the beginning of the simulation period in 2004.

Treated areas had significantly greater basal area for at least four decades (Fig. 4). After 100 years, total basal area was very similar under all treatment and regeneration combinations. Trees in untreated areas were larger on average immediately post-fire due to greater mortality of small trees compared to treated areas, but because of more regeneration in untreated areas, average tree diameter soon dropped below that of treated areas (Fig. 4). As with tree density, this difference lessened over time and, by the end of the simulation, treated and untreated areas were nearly the same, with a slightly larger average diameter under Regen-1.

In both treated and untreated areas, an open forest with some surviving ponderosa pines and a few survivors of other species quickly gave way to a thicket of oaks, junipers, and New Mexico locust, with the occasional remnant ponderosa pine (Fig. 5) making up only around 5% of total density under Regen-1 and 10% under Regen-2. In treated areas, about a quarter of the trees were ponderosa pines after the 2004 regeneration became established, and this proportion remained stable over time; treated areas also had many more ponderosa pines than untreated areas in absolute numbers. Two of the seven untreated sites had only 21 and 26 ponderosa pines ha^{-1} by the end under Regen-1. The relative density of New Mexico locust declined slowly, from 15 to 30% at its maximum, to 10–18% after 100 years, while the proportion of oaks increased slightly over time. About half the trees were oaks in untreated areas, and 35% in treated areas. The proportion of junipers was roughly the same in all treatment and regeneration combinations – around 25% – and remained steady over the course of the simulation.

Basal area differences between treated and untreated areas became greater over time with little difference between the regeneration scenarios (Fig. 5). Ponderosa pine initially made up all the basal area in untreated areas, but after four decades, treated areas had proportionally more ponderosa pine. After 100 years, treated areas were ~60% ponderosa pine, while untreated areas were down to 35%. Oak basal area was largely responsible for the similarity in total basal area between treated and untreated areas by the end of the simulation. In treated areas, oaks originally constituted about 5% and never went beyond 20% of total basal area, whereas in untreated areas they comprised 40% of the basal area after 100 years.

Discussion

Despite displaying some of the most intense fire behaviour and covering the largest severely burned landscape ever observed in the South-west, the Rodeo–Chediski fire still decreased in

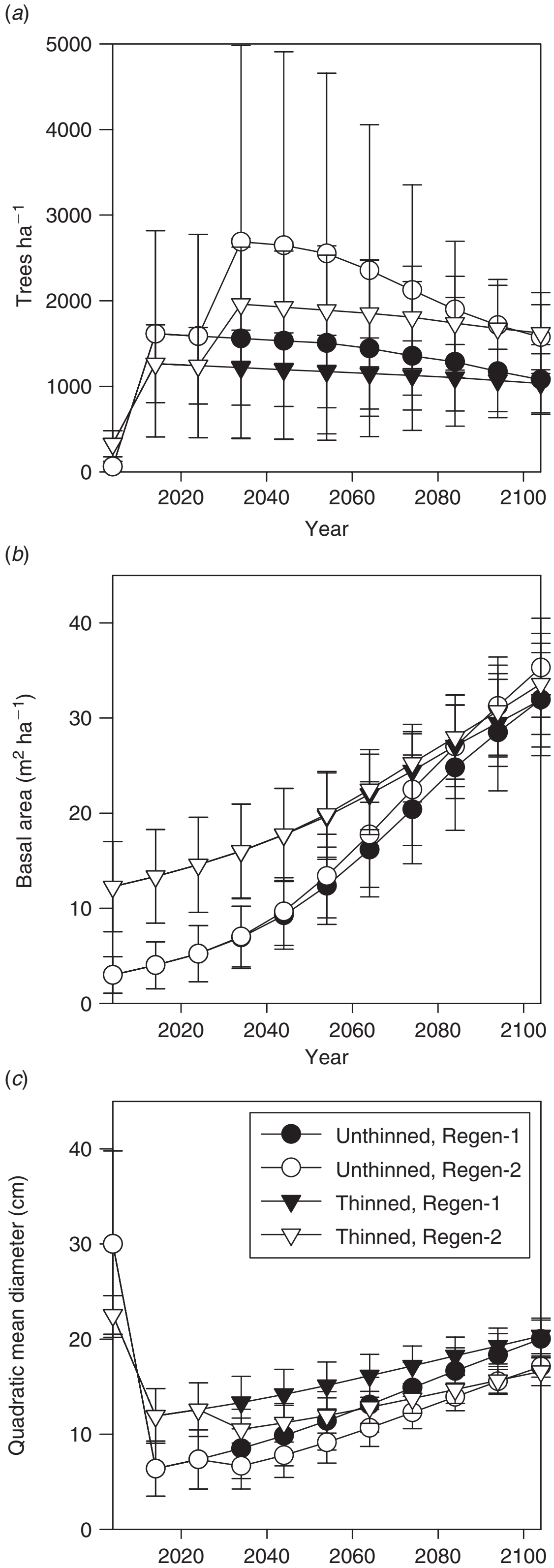

Fig. 4. Simulation modelling forecasts that untreated sites quickly become denser than treated sites due to more dense regeneration. (*a*) Density, (*b*) basal area and (*c*) quadratic mean diameter.

severity in areas that underwent fuel reduction treatments before the fire. Treated areas were significantly different from untreated areas in terms of forest structure and some attributes of initial post-fire recovery. We expect future forest development to differ as well, especially in terms of dominance by ponderosa pine.

Sampling 2 years post-wildfire allowed us to estimate future forest development from a relatively accurate starting point. McHugh and Kolb (2003) ascertained that most mortality after wildfire in northern Arizona ponderosa pine forests occurs within the first 2 years, so our measurements of survival should be reliable. Regeneration had become well established after the passage of two growing seasons, so our projections of future growth are based on more complete data than could be gathered very soon after the fire.

The disadvantage to deferring post-fire sampling is that fire-killed trees may fall. Because we used variable-radius plots, which are not suited to measuring fallen trees, we do not have a complete sample of trees that had already fallen after being killed by the fire. However, we rarely encountered already-fallen trees in the study area and most fire-killed ponderosa pine stand $\geq$3 years (Chambers and Mast 2005), so we believe our estimates of pre-fire density to be reasonably accurate.

Pre-wildfire treatment effects

Treated areas experienced significantly less bole char and reduced tree mortality than untreated areas. This is in accordance with previous studies indicating the effectiveness of fuel reduction treatments or prescribed burning alone at reducing wildfire severity (Pollet and Omi 2002; Cram and Baker 2003; Martinson and Omi 2003; Finney *et al.* 2005). Overstorey survival was significantly greater: half the trees in treated areas survived, compared to 5% in untreated areas. After the fire, there were more live trees in terms of density and basal area in treated areas, and the diameter distribution did not substantially change in shape. Some portions of treated as well as untreated areas still experienced complete overstorey mortality, but this was limited to 11% of treated plots as opposed to 66% of untreated plots. The fire raised crown base height in untreated areas, but not in most portions of treated areas. While our statistical tests did not indicate that pre-fire forest structure on treated and untreated sites was significantly different, average overall tree density and density of small trees on untreated sites was considerably greater than on treated sites. When compared to the treatments tested by Pollet and Omi (2002), our ratio of average densities in pre-fire forest structure between treated and untreated areas was within the range of treatments they also found effective, albeit on the low end. Our untreated sites averaged 1.7 times denser than our treated sites; the untreated sites they studied had between 1.4 and 8.7 times more trees than their treated sites. Pre-fire basal area was nearly identical on our treated and untreated sites, which Pollet and Omi (2002) also found to be the case on two of the four wildfires they sampled. These findings suggest that fire severity is coupled more closely to the arrangement of tree fuels – distributed among many smaller trees *v.* fewer, larger ones – than to the total amount of tree fuels.

Future forest development

Regeneration 2 years after the fire was strongly dominated by Gambel oak, alligator juniper, and New Mexico locust, making

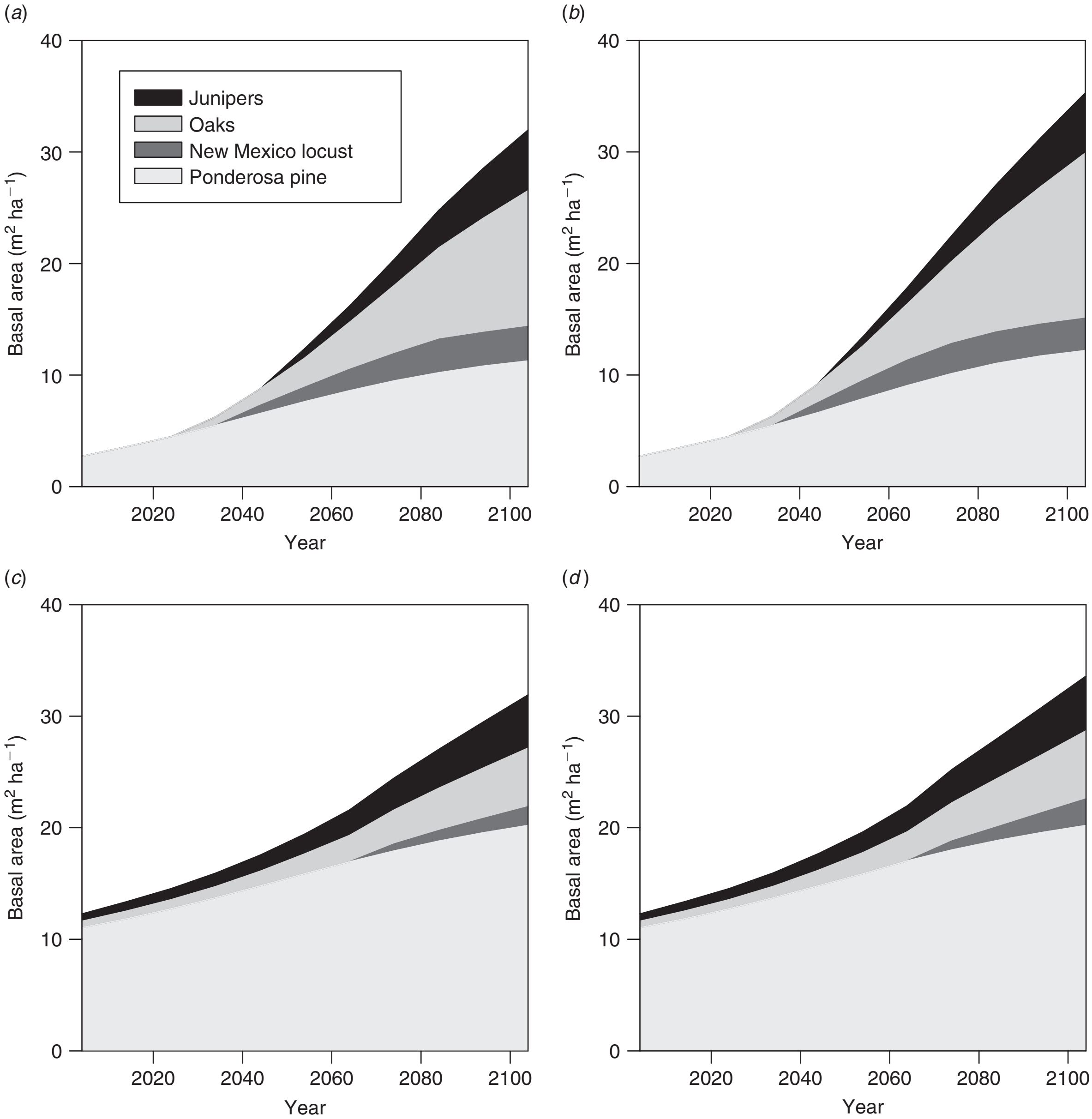

Fig. 5. Simulation of basal area change by species shows pine dominance throughout the next 100 years in the treated sites under both regeneration scenarios. Untreated sites, in contrast, become dominated by non-pine species. Unthinned (*a*) Regen-1 and (*b*) Regen-2; thinned (*c*) Regen-1 and (*d*) Regen-2.

it very different from the ponderosa pine-dominated overstorey. Our modelling results indicate that treated areas are likely to be less dense, have larger trees on average, and have greater basal area than untreated areas for the next several decades. Both treated and untreated sites became a thicket of young oaks, junipers, and New Mexico locust, with both remnant and young ponderosa pines more common on treated sites but still not comprising more than a quarter of all trees. By 100 years post-fire, stands were projected to reach high levels of density and basal area. These results may not materialise due to uncertainties in the model algorithms, as well as the differences between modelled and actual climate during the next century, discussed further below, but the upper limits of tree densities and basal area do not exceed levels observed in some contemporary Arizona ponderosa pine forests (Covington *et al*. 1997; Fulé *et al*. 2001).

Despite the lack of a significant difference in overall regeneration between treated and untreated sites as measured in 2004, the slightly greater regeneration levels on untreated sites led to their greater tree density over the course of the simulation, especially under the second regeneration scenario. Ponderosa pines still constituted the majority of basal area in treated areas after 100 years, but accounted for only 35% of total basal area in untreated areas. The proportion of oaks in terms of both density and basal area increased over time in treated and untreated areas.

The high variability in model outputs, illustrated in the error bars in Fig. 4, might be expressed in terms of a patchy landscape with pine groups scattered in a matrix of shrubfields.

Savage and Mast (2005) delineated several trajectories ponderosa pine forests have taken after crownfire: burned landscapes have turned back into dense pine forests, become early-successional ponderosa pine/Gambel oak forests, or undergone type conversions into potentially self-perpetuating grasslands or oak/manzanita shrubfields. Since some regeneration was present on every site and 86% of the plots, conversion to grassland is unlikely. Sixty-four percent of untreated and 11% of treated plots had no surviving pines and no ponderosa pine regeneration; conversion to an oak/manzanita shrubfield is more likely in these areas. In both treated and untreated areas, future growth will in general be a combination of oaks, junipers, pines, and New Mexico locust, rather than the strongly ponderosa pine-dominated pre-wildfire forest.

It is uncertain how long an altered successional trajectory such as an oak/manzanita shrubfield may persist, and what stage might occur next (Barton 2002). Savage and Mast (2005) found that these conditions could persist for at least five decades, and pointed out that these type conversions may be self-perpetuating if the area repeatedly experiences intense fire. Tree planting can substitute for natural regeneration, but Savage and Mast (2005) cautioned that many post-wildfire plantations of ponderosa pines have failed. On the other hand, a shrubfield near an area with surviving pines may eventually revert to a ponderosa pine forest (Savage and Mast 2005). Our plots with complete mortality of ponderosa pine trees and no ponderosa pine regeneration (38% of the plots overall) were patchily distributed across the study sites, so nearby trees could serve as seed sources. However, even if these areas ultimately become ponderosa pine forests again, the recovery process would most likely take centuries rather than decades. During the interim, ponderosa pine forests on the burn area would be fragmented and considerably reduced in total size.

Climate change is projected to accelerate shifts in vegetational states caused by fire. While site-specific forecasts remain uncertain, average temperature and precipitation are both expected to increase in the western USA and Canada, extending the fire season and increasing both fire frequency and total area burned because of the greater amplitude and duration of extreme fire weather (Flannigan *et al.* 2001; Westerling *et al.* 2006). In the South-west, the largest fires are associated with a current-year drought after several years of above-average precipitation (Swetnam and Betancourt 1990; McKenzie *et al.* 2004), so higher average precipitation may actually increase the number of large fires. This effect of climate change on fire regimes may indirectly hasten vegetation shifts more than the direct effect of climate change itself on vegetation; burn areas offer sites for invasive species such as *Bromus tectorum* to colonise, and increased fire size and frequency may decrease the habitat available to late-successional species (Flannigan *et al.* 2001; McKenzie *et al.* 2004). Old ponderosa pine forests, already rare, may become further imperilled.

Management implications

The Rodeo–Chediski fire merits attention not only for its prominence in eastern Arizona but also because similar post-fire landscapes are likely to dominate increasingly larger fractions of the South-west's forests. The unintended long-term effect of our history of fire suppression, overgrazing, and conventional timber management focussing on the harvesting of large trees may be the increasing fragmentation of the largest contiguous ponderosa pine forest in the United States, with resultant long-term soil damage, loss of timber revenue, and ecological and social effects that are not yet entirely known. Climate change is forecasted to accelerate these shifts.

Even under the extreme conditions of drought and wind that characterised the Rodeo–Chediski, however, fuel reduction treatments that took place before the fire had a major effect in reducing fire severity and helping perpetuate ponderosa pine ecosystems. These treatments included prescribed burning (Finney *et al.* 2005) and thinning (Cram and Baker 2003; this study). These treatments comprised only a small portion of the burn area, however (Finney *et al.* 2005).

Field data and simulation results indicate that ponderosa pine will lose dominance for the next several decades to centuries over a considerable proportion of the burn area, to be replaced by a matrix of oak/manzanita shrubfields and thickets of oaks, junipers, and New Mexico locust with the occasional remnant ponderosa pine. In contrast, most treated areas should recover to a ponderosa pine/Gambel oak forest relatively quickly. While tree planting is costly and often fails in the South-west, it should be considered because of sparse ponderosa pine regeneration and the likelihood of shrub species dominance, especially in untreated areas.

Most south-western forests are currently at high risk of severe crownfire (Fiedler *et al.* 2002; Schmidt *et al.* 2002). Unless extensive fuel reduction treatments are undertaken in the near future, preferably in ways that integrate strategic fuelbreak locations (Finney 2001) and conservation priorities (Noss *et al.* 2006), many of these forests will experience severe fires. In 1951, Weaver recommended thinning of small-diameter trees and the use of prescribed fire to improve forest health and prevent severe wildfire in ponderosa pine forests (Weaver 1951). Forest management involving fuel reduction treatments in an ecological restoration context are now also recommended by many others (Covington *et al.* 1997; Agee and Skinner 2005). At least in strategic areas, treatments such as thinning and prescribed burning should help ponderosa pine forests persist even after extraordinary fires, such as the Rodeo–Chediski.

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