Forest Ecology and Management 297 (2013) 75–83

Contents lists available at SciVerse ScienceDirect

Forest Ecology and Management

journal homepage: www.elsevier.com/locate/foreco

# Pre-wildfire management treatments interact with fire severity to have lasting effects on post-wildfire vegetation response

Kristen L. Shive [a,*], Carolyn H. Sieg [b], Peter Z. Fulé [a]

[a] School of Forestry, Northern Arizona University, P.O. Box 15018, Flagstaff, AZ 86011, USA
[b] Rocky Mountain Research Station, 2500 S. Pine Knoll Dr., Flagstaff, AZ 86001, USA

ARTICLE INFO

*Article history:*
Received 1 October 2012
Received in revised form 13 February 2013
Accepted 14 February 2013
Available online 19 March 2013

*Keywords:*
High severity
Ponderosa pine
Pine regeneration
Exotic species
Arizona
Rodeo-Chediski Fire

ABSTRACT

Land managers are routinely applying fuel reduction treatments to mitigate the risk of severe, stand-replacing fire in ponderosa pine communities of the southwestern US. When these treatments are burned by wildfire they generally reduce fire severity, but less is known about how they influence post-wildfire vegetation recovery, as compared to pre-fire untreated areas. We re-measured existing plots on the 2002 Rodeo–Chediski Fire 8 years after the wildfire to track plant community and exotic species response, as well as patterns of pine regeneration. We compared areas that experienced high- and low-severity burning, and also examined how pre-fire treatment (cutting in an uneven-aged harvesting system with prescribed fire) modified vegetation response. We detected persistent differences between low- and high-severity areas for nearly all variables measured. In high-severity areas overall understory plant cover was 40.6%, nearly three times that observed in low-severity areas; shrub cover was 18.4%, four and a half times greater than that observed in low-severity areas. We also detected significantly higher exotic forb cover in high-severity areas, although overall exotic response was generally quite low (<2%). Although this represents a slight decrease in exotic cover since the initial 2004/2005 measurements, the frequency of several exotic species did increase through time (particularly *Tragopogon dubius* and *Verbascum thapsus*). Pre-fire treatment resulted in significantly higher pine regeneration frequency in treated versus untreated areas. Within low severity areas, mean pine regeneration frequency was 0.17 in pre-fire untreated areas versus 0.06 in areas that were not treated before the fire. Within high severity burned areas, mean pine regeneration frequency was 0.67 in pre-fire treated areas, but was only 0.19 in pre-fire untreated areas. This treatment effect in high-severity areas may be linked to reduction in the overall patch size of high burn severity in pre-fire treated areas, which resulted in a more heterogeneous mixture of low and moderate severity burning in the neighborhood. This pattern decreased distance to seed source, which likely facilitated the more frequent pine regeneration observed. In addition to the well-documented benefits of fuel reduction treatments in reducing subsequent fire severity, these data suggest that even where treated areas do burn severely the size of severely burned patches is limited in extent, which is likely to have important ramifications for future reforestation and retention of foundation species.



## 1. Introduction

Severe large-scale fires in ponderosa pine (*Pinus ponderosa* P. & C. Lawson) forests of the southwestern US have been linked to uncharacteristically high fuels (Covington and Moore, 1994) and climate warming in recent decades (Westerling et al., 2006). These fire events have raised several important concerns about post-fire vegetation communities, three of which we examine here. First, extensive forest areas that have burned severely may result in persistent conversions to shrublands or grasslands, rather than regenerate as forest (Haire and McGarigal, 2008; Savage and Mast, 2005), particularly under a warmer and drier climate (Seager et al., 2007). The unprecedented size of high-severity burned areas necessarily means that post-fire pine regeneration in these areas is not well understood, because it is occurring in relatively novel conditions. In addition, high-severity fire may promote sprouting species over seeding species (such as ponderosa pine), where early site capture by sprouting species limits seeder species establishment through competition for resources (Barton, 2002; Fulé and Covington, 1998). Furthermore, ponderosa pine can be considered a foundation species in these communities because it defines community structure by creating locally stable conditions for other species, and modulates fundamental ecosystem processes. The loss of a foundation species over large, landscape-scale areas can create

* Corresponding author. Tel.: +1 928 523 3031; fax: +1 928 523 1080.
*E-mail addresses:* kls448@nau.edu (K.L. Shive), csieg@fs.fed.us (C.H. Sieg), Pete.Fule@nau.edu (P.Z. Fulé).

0378-1127/$ - see front matter © 2013 Elsevier B.V. All rights reserved.
http://dx.doi.org/10.1016/j.foreco.2013.02.021

novel species assemblages and degrade native habitats (Ellison et al., 2005).

Second, numerous studies have documented exotic plant species invasions following high severity fire events in the Southwest (Barclay et al., 2004; Crawford et al., 2001; Dodge et al., 2008; Griffis et al., 2001). Such invasions can impact native plant communities, and also have the potential to alter future fire regimes via changes to the fuels properties of the native ecosystem (Brooks et al., 2004). These invasions are not entirely predictable, however, since other high severity fire studies in the Southwest have documented a low exotic species response (Foxx, 1996; Huisinga et al., 2005; Kuenzi et al., 2008).

Third, the extent to which pre-fire fuels treatments will influence post-fire vegetation response is of concern as land managers are routinely applying fuel reduction treatments (e.g., thinning and/or burning) to increase forest resilience and reduce high-severity fire risk (Baron et al., 2009). These treatments generally reduce fire severity by reducing surface fuels, and reducing tree density and canopy connectivity (Cochrane et al., 2012; Finney et al., 2005; Fulé et al., 2012; Pollet and Omi, 2002; Strom and Fulé, 2007), yet the influence of extreme fire weather conditions and topography can occasionally result in high-severity fire in treated areas. In both lesser- and more-severely burned sites, pre-fire treatments are likely to have long-term effects on post-fire recovery, yet these effects are not well known.

To investigate the longer-term effects of pre-wildfire treatments and fire severity on pine regeneration and native plant community recovery, we re-measured plots established after the 2002 Rodeo–Chediski Fire of northeastern Arizona that burned approximately 189,650 ha (Kuenzi et al., 2008; Strom, 2005). These plots were originally installed in 2004, when herbaceous understory, fuels, and forest structure response variables were measured in relation to fire severity and pre-fire treatment. Kuenzi et al. (2008) found that low- and high-severity areas had markedly different plant communities 2 and 3 years post-fire, which they primarily attributed to the near-complete loss of overstory trees in high-severity areas (Kuenzi et al., 2008). These early differences indicated two drastically different vegetation types on the landscape: in the absence of another high-severity fire event, low-severity areas are likely to remain pine-dominated systems, whereas high-severity areas have the potential to convert to other vegetation types (Haire and McGarigal, 2010; Savage and Mast, 2005; Strom and Fulé, 2007). They also detected a low exotic species response, contrary to expectations.

Eight years post-fire, we re-visited these plots to examine the influence of fire severity and pre-fire treatment on the post-fire vegetation community. We asked: (1) are the differences in low- versus high-severity areas persistent 8 years post-fire? And (2) do pre-fire treated and untreated areas differ within low severity or high severity areas? For each line of inquiry, we examined a suite of characteristics that pertain to forest recovery: understory plant community composition (with particular emphasis on exotic species response), overstory structure, and pine regeneration.

## 2. Materials and methods

### 2.1. Study area

The study area lies south of the Mogollon Rim in east-central Arizona on White Mountain Apache Tribal (WMAT) lands. For the 8 years of post-fire vegetation recovery, precipitation by water year was generally below the 60-year average of 45.0 cm. Total precipitation during 2010 was 35.2 cm, and 99.8 cm of snow accumulated over the winter of 2009/2010, which is only slightly below the 60-year average of 102.4 cm. Average maximum and minimum monthly temperatures were 29.2 °C in July and −8.7 °C in January. Weather data were recorded at the Heber Ranger Station by the Western Regional Climate Center (www.wrcc.dri.edu), which is located above the Rim at an elevation of 1984 m (16–311 m lower than the range of our study sites), approximately 12 km north of the study sites. It is on the edge of a ponderosa pine/pinyon-juniper transition zone and so may have recorded less precipitation and higher temperatures than our study sites experienced.

The study sites were randomly selected in 2004, ranging in elevation from 2000 to 2295 m, on slopes <45% (average slope was 17.6%) (Kuenzi et al., 2008; Strom, 2005). Study sites were stratified by fire severity classes (low and high), and by pre-wildfire management of (1) no treatment or (2) cut in an uneven-aged silvicultural system and subsequently burned under prescription (hereafter: untreated, treated) within 11 years prior to the fire. The specific cutting prescriptions were not uniform across the study area, but generally included uneven-aged forest management and noncommercial thinning followed by slash disposal. Uneven-aged managements followed a range of a *Q*-slope factor of 1.1–1.3, an SDI maximum of 1110 25.5-cm trees $ha^{-1}$, and maximum diameter of 46–76 cm (18–30 in). Out of the approximately 111,837 ha of the Rodeo–Chediski Fire that burned on WMAT lands, 13% were cut and burned during the time period used for this analysis (11 years) (Strom, 2005). Pre-fire treatment areas were identified using treatment history data provided by the tribe. Fire severity classes were determined by a combination of remotely sensed Differenced Normalized Burn Ratio (ΔNBR) maps and ground-truthing. Of the 111,837 ha WMAT fire area, nearly 20% was in the high-severity class and 29% in the low-severity class (with the remaining area classified as "moderate" severity) (Strom, 2005). The sites were randomly distributed throughout the treatment/severity combinations, resulting in distances between sites ranging from 0.55 to 9.37 km, and averaging 1.66 km.

The high-severity study areas were seeded immediately post-fire. The seed mix used on the WMAT lands included *Triticum aestivum* L. (common wheat), a sterile exotic species, which was applied at a rate of 16.8 kg $ha^{-1}$. The remainder of the seed mix included species native to the region, but the seed source is unknown and therefore may not represent locally native genotypes: *Elymus trachycaulus* (Link) Gould ex Shinners (slender wheatgrass), *Pascopyrum smithii* (Rydb.) A. Löve (western wheatgrass), *Panicum virgatum* L. (switchgrass), *Nassella viridula* (Trin.) Barkworth (green needlegrass), *Bromus marginatus* Nees ex Steud. (mountain brome) (*Bromus marginatus* = *Bromus carinatus* H. & A. (Welsh et al., 1993)), *Bouteloua curtipendula* (Michx.) Torr. (sideoats grama), *Sporobolus cryptandrus* (Torr.) Gray (sand dropseed), *Coreopsis tinctoria* Nutt. (plains coreopsis), *Dalea purpurea* Vent. (purple prairie clover), *Linum lewisii* Pursh (blue flax), and *Rudbeckia hirta* L. (black-eyed susan) (Kuenzi et al., 2008). The study area was not in an active cattle allotment, but we did observe a few horses, and evidence of deer and elk.

### 2.2. Measurement

Six sites were originally installed within each of four treatment and severity combinations (total $N = 24$), with three subsample plots at each site. In 2010 we re-located 70 of the 72 original plots, such that two sites had only two subsample plots. For consistency, all field protocols follow the original studies (Kuenzi et al., 2008; Strom, 2005). We used a variable-radius prism plot (basal area factor 2.3 $m^2$ $ha^{-1}$) for overstory trees using standard guidelines (Avery and Burkhart, 2001), in which we measured diameter at breast height (DBH, 1.37 m) and tree height. We tallied all pine regeneration shorter than breast height in a 3.1-m radius circle. Since smaller seedlings generally have high mortality rates (Smith et al., 1997), we were interested in separating the more established

seedlings from the very small seedlings, and so divided them into two height classes, 0–40 cm and 41–137 cm. We used a hemispherical fisheye lens (FC-E8 Fisheye Converter Lens) attached to a Nikon CoolPix E4300 digital camera to photograph canopy cover, and then quantified canopy openness using the Gap Light Analyzer (Institute of Ecosystem Studies, 1999). In addition, we re-took the plot overview photos in order to create a photograph series through time.

We measured understory characteristics using two perpendicular, 44.8-m transects to create the 1000-$m^2$ plot. Along each transect, we measured plant canopy cover using cover classes that were adapted from Daubenmire (1959) (<1%, 1–5%, 6–25%, 26–50%, 51–75%, 76–95%, 96–100%) in ten 20- × 50-cm quadrats, for a total of 20 quadrats per plot. We recorded cover data by life form and nativity as well as by species, including trees shorter than 1.37 m, and recorded a species list for the entire plot area.

To directly compare our data, we followed Kuenzi et al.'s (2008) taxonomic and nativity classification which was based on the USDA-NRCS PLANTS database (USDA-NRCS, 2006). One exception they made was *Portulaca oleracea* L. (little hogweed), which was listed as "introduced" on the PLANTS database, but Kuenzi et al. (2008) classified it as "native" based on Byrne and McAndrews (1975). For species that were first detected in 2010, we used the more current USDA-NRCS PLANTS database (USDA-NRCS, 2011). When we were unable to identify plants to species, we classified them to the lowest taxonomic level we could identify with certainty, generally to the genus level.

Our preliminary analyses suggested there were some differences in vegetation response between treated and untreated high-severity areas, and we suspected that pre-fire treatment may have limited overall high-severity patch size. Because of the loss of most overstory trees in high-severity areas, high-severity patch size can increase distance to seed source, which can be a major factor influencing patterns of seedling recruitment and re-forestation. Therefore, we examined spatial patterns of fire severity in the neighborhood surrounding our high-severity sampling sites. To do this, we defined the potential seed contribution area for a site based on seed dispersal distances, which have been estimated at 1.5 times tree height (McDonald, 1980). Since there is some variability in the literature regarding the minimum seed tree size (Larson and Schubert, 1970; Pearson, 1950) we selected an intermediate DBH of those reported, 40.6 cm (16 in). We calculated the average height of trees this size or larger across our entire study area and calculated the estimated dispersal distance to be approximately 36 m. We then generated a buffer of at least 36 m from any subsample plot at each study site, which resulted in an overall circular site buffer of 180 m from site center (Fig. 1). Then we used the dNBR severity map to count the number of low, moderate and high-severity pixels within the potential seed contribution area, and averaged these by treatment. For consistency, all pixels that partially intersected the buffer were also included in the pixel counts.

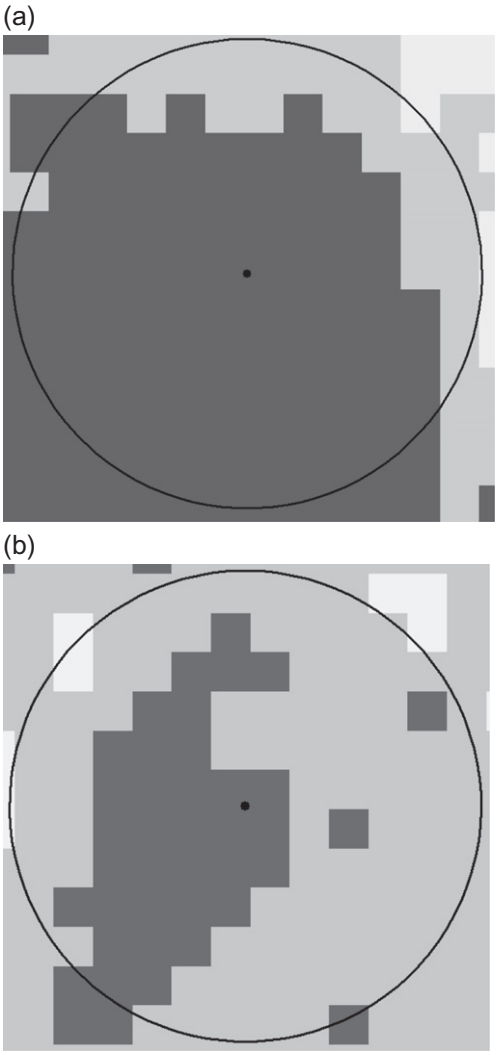


**Fig. 1.** Example of how neighborhood severity analysis was conducted on pre-fire untreated (a) and treated (b) sites. The dot represents site center, and the 180 m circle represents the area of potential seed contribution. The darkest shade of grey represents high severity, the lighter shade represents moderate severity and the lightest shade represents low severity.

### 2.3. Statistical methods

For all analyses, we used site level means ($n = 24$). We analyzed plant community data using a combination of univariate and multivariate statistical tests as well as ordination techniques. All plant community analyses were based on the subset of species that occurred in at least 5% of the plant cover quadrats, and richness analyses were based on species that occurred on at least 5% of the plot species lists, in order to reduce the influence of uncommon species (McCune and Grace, 2002). For overall plant community composition we used PC-ORD software (McCune and Mefford, 1999) to perform two-way PerMANOVA tests, a non-parametric permutation procedure, on burn severity (high, low) and pre-fire treatment (untreated, treated). We used the Bray-Curtis dissimilarity measure with 9999 permutations, with $\alpha = 0.05$. We analyzed differences among univariate response variables such as cover by life form and nativity, species richness and pine regeneration relative frequency with PerMANOVAs in PC-ORD, using Euclidean distance (Anderson, 2001). When the interaction term was significant, we conducted subsequent perMANOVAs for treatment within each severity class.

We then graphically displayed community data by severity, treatment, and severity/treatment combinations in ordination space, using non-metric multidimensional scaling (NMDS). We conducted 250 runs with real data and 250 runs on randomizations from a random starting point, with an instability criterion of 0.00001. We chose the final number of axes based on stress levels <15 with $p$-values < 0.05 (Peck, 2010). Finally, we also used PC-ORD to identify indicator species for severity, treatment, and severity/treatment combinations. Indicator species analysis yields a value based on relative frequency and percent cover for each species. Species with an indicator value >25 and a $p$-value (based on a Monte Carlo test) of <0.05 were considered indicator species for that severity class (Dufrene and Legendre, 1997). To examine

patterns of pine regeneration we focused on frequency, or the proportion of plots with at least one seedling out of the total number of plots. Finally, to test for differences in neighborhood severity for the high-severity sites, we grouped the low- and moderate-severity pixel percentages, since we considered both of these severity classes as a proxy for seed sources due to overstory tree retention. Using the Shapiro–Wilk test we found that these data were normally distributed ($p = 0.3989$), and so we tested the differences in the proportion of low/moderate severity area between treated and untreated areas with a t test (both tests performed with JMP, Version 7).

## 3. Results

We documented the presence of a total of 329 species from 190 genera and 60 families. For multivariate plant community analyses on cover we focused on the 104 species found on at least 5% of the cover quadrats, and for richness analyses we focused on the 172 species found on at least 5% of the plot census data. There was a total of 22 exotic species from eight families, most from Poaceae (9) and Asteraceae (5); 10 of these were found on at least 5% of the plots (Table 1).

### 3.1. Overstory structure

Overstory structure variables were all significantly different ($p < 0.001$) between low- and high-severity areas, where low-severity areas had higher basal area and tree density, and lower canopy openness, due to the loss of most overstory trees in high-severity areas. None of the forest structure variables were significantly different by treatment (Fig. 2 and Table 2). In spite of the general lack of significant effects, the mean responses in low-severity areas suggest a more open structure in treated compared to untreated sites, where there was lower average tree density and basal area and higher mean percent canopy openness. In high-severity areas, the differences in mean tree density and basal area in treated and untreated sites were negligible (Fig. 2 and Table 2).

### 3.2. Plant community composition

Plant community composition based on cover by species had a significant severity by treatment interaction ($p = 0.019$). Subsequent tests for treatment effect within severity class were significant both for both low ($p = 0.047$) and high ($p = 0.017$) severity. Within low severity, *Solidago* spp. were an indicator species for the treated areas and no indicator species were detected for untreated areas. Within high severity, *Muhlenbergia virescens* Hitchc. (screwleaf muhly) and ponderosa pine (regeneration under breast height) were indicator species of the treated areas. In addition, two species were much more strongly represented in the high-severity untreated areas than the high-severity treated areas, where untreated areas had 42 times more cover of the perennial grass *P. smithii* and nine times more cover of the shrub-like tree, *Robinia neomexicana* A. Gray (New Mexico locust) than the treated areas.

The NMDS ordination of high severity sites generally grouped by treatment, but the low-severity communities were too variable to detect any distinct pattern by treatment (figures not shown). We also examined NMDS ordinations and indicator species for severity and treatment alone. The NMDS ordination by severity alone (not shown) mimicked the results presented in Kuenzi et al. (2008), where plant communities generally grouped by severity class, and the overstory structure vectors indicated that these groupings were driven by the overstory structure differences between low and high severity.

When the ordinations were generated by treatment alone, distinct groupings were not visible. Indicator species of high severity include three shrub species: *Arctostaphylos pringlei* Parry (Pringle manzanita), *Arctostaphylos pungens* Kunth (pointleaf manzanita) and *Ceanothus fendleri* A. Gray (Fendler's ceanothus). Other indicator species for high severity included three native graminoid species seeded post-fire: *P. smithii*, *Panicum virgatum*, and *B. curtipendula* (Table 3). No indicator species were detected for low-severity areas or by treatment due to highly variable plant community composition.

Species richness was significantly higher in high-severity areas than low-severity areas ($p = 0.007$) (Fig. 3 and Table 2). This was

**Table 1**
Exotic graminoid and forb species by burn severity class. Seeded species include species known to have been seeded on the WMAT lands, not necessarily after the Rodeo–Chediski Fire (Kuenzi et al., 2008). Bolded species were found on at least 5% of plots.

| Graminoids | | | Forbs | | |
|---|---|---|---|---|---|
| Severity | Seeded | Other | Severity | Seeded | Other |
| **High** | *Agropyron desertorum*[a]<br>*Dactylis glomerata*[a]<br>***Eragrostis curvula***[a] | *Bromus japonicus*[a]<br>***Bromus tectorum***[a]<br>*Echinochloa crus-galli*<br>***Poa compressa***<br>***Poa pratensis***[a]<br>*Setaria viridis* | **High** | None | *Cirsium vulgare*[a]<br>***Erodium cicutarium***[a]<br>***Lactuca serriola***[a]<br>*Malva neglecta*<br>***Medicago lupulina***[a]<br>*Melilotus officinalis*[a]<br>*Onopordum acanthium*<br>*Polygonum aviculare*[b]<br>*Rumex acetosella*<br>***Taraxacum officinale***<br>***Tragopogon dubius***<br>***Verbascum thapsus*** |
| **Low** | *Dactylis glomerata*[a]<br>***Eragrostis curvula***[a] | ***Bromus tectorum***[a]<br>***Poa compressa***<br>***Poa pratensis***[a] | | | |
| | | | **Low** | None | *Cirsium vulgare*[a]<br>***Erodium cicutarium***[a]<br>***Lactuca serriola***[a]<br>***Medicago lupulina***[a]<br>*Melilotus officinalis*[a]<br>*Rumex acetosella*<br>*Rumex crispus*<br>*Sanguisorba minor*<br>***Verbascum thapsus*** |

[a] Species that have been seeded as part of management practices in the Southwest (Fowler et al., 2008; Kuenzi et al., 2008).

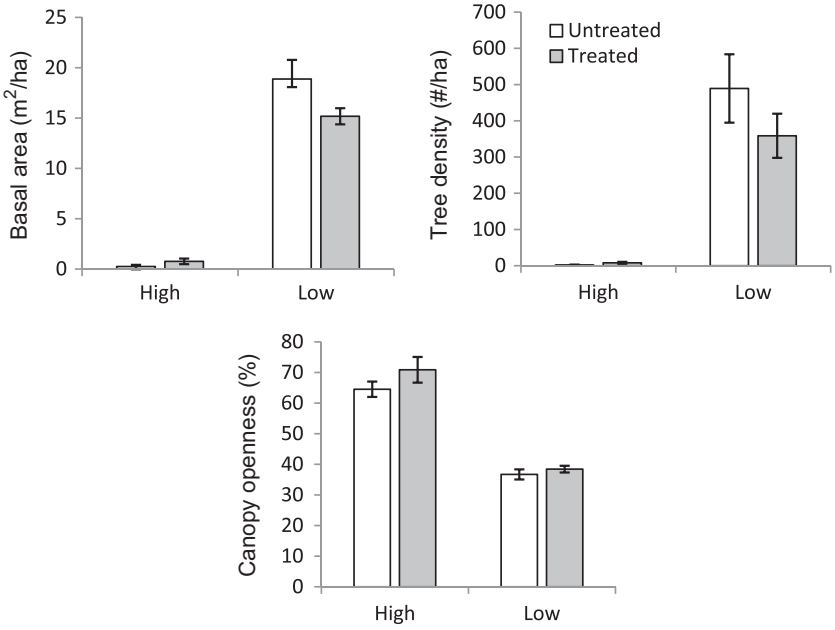


**Fig. 2.** Overstory variables: basal area, tree density, canopy openness by treatment and severity, with standard error. All were significantly different by severity class (high and low), but none were significantly different by treatment within severity class.

**Table 2**
*P*-values from perMANOVA tests on univariate response variables. Significance at $\alpha = 0.05$ is denoted by bold type.

| Response variable | Severity | Treatment | Interaction |
|---|---|---|---|
| Overstory tree density | **<0.001** | 0.312 | 0.266 |
| Basal area | **<0.001** | 0.143 | 0.059 |
| Canopy openness | **<0.001** | 0.144 | 0.385 |
| Total plant cover | **<0.001** | **0.019** | **0.019** |
| Tree regeneration cover | **0.005** | 0.454 | **0.029** |
| Shrub cover | **<0.001** | 0.070 | 0.324 |
| Forb cover | **0.002** | 0.450 | 0.075 |
| Graminoid cover | **0.008** | 0.574 | 0.143 |
| Exotic forb cover | **0.004** | 0.967 | 0.912 |
| Exotic graminoid cover | 0.649 | 0.231 | 0.157 |
| Species richness | **0.007** | 0.198 | 0.630 |
| Exotic species richness | 0.070 | 0.364 | 0.141 |
| Native species richness | 0.096 | 0.608 | 0.329 |
| Pine regeneration (41–137 cm height class) frequency | **0.006** | **0.008** | 0.079 |

**Table 3**
High severity indicator species. No indicator species were found for the low-severity sites. Exotic species are in bold. Indicator species for severity/treatment combinations are noted in the text.

| Species | Indicator Value (IV) | *P*-value | Mean cover | S.E. |
|---|---|---|---|---|
| *Arctostaphylos pringlei* | 46.7 | 0.036 | 2.87 | 0.83 |
| *Arctostaphylos pungens* | 55.6 | 0.032 | 3.29 | 0.95 |
| *Bouteloua curtipendula*[a] | 73.1 | 0.004 | 1.55 | 0.45 |
| *Carex* sp. | 65.3 | 0.024 | 1.03 | 0.30 |
| *Ceanothus fendleri* | 79.7 | 0.004 | 14.25 | 4.11 |
| *Cirsium wheeleri* | 84.5 | 0.004 | 0.81 | 0.23 |
| *Dyssodia papposa* | 41.7 | 0.036 | 0.06 | 0.017 |
| *Lotus wrightii* | 87.9 | 0.004 | 0.55 | 0.16 |
| *Panicum virgatum* | 41.7 | 0.048 | 0.30 | 0.09 |
| *Pascopyrum smithii*[a] | 74.9 | 0.004 | 3.20 | 0.92 |
| *Robinia neomexicana* | 65.6 | 0.004 | 4.25 | 1.23 |
| ***Tragopogon dubius***[b] | 75.0 | 0.004 | 0.05 | 0.016 |
| ***Verbascum thapsus***[b] | 73.8 | 0.016 | 0.55 | 0.16 |

[a] Seeded.
[b] Exotic.

driven primarily by greater forb richness in high-severity areas, where approximately 37 forb species were observed, versus 31 forb species in low-severity areas. Species richness was not significantly different by treatment ($p = 0.198$), but the means were slightly higher in treated areas. For total understory plant cover, the interaction term was significant ($p = 0.019$). In high severity areas, we detected significantly higher total understory plant cover in the untreated sites as compared to the treated sites ($p = 0.033$). In low severity areas, this trend was reversed although that difference was not significant ($p = 0.247$).

### 3.3. Cover by life form and nativity

Tree regeneration, shrub, forb and graminoid cover were all significantly higher in high-severity areas as compared with low-severity areas (Table 2 and Fig. 3). Shrub cover was a substantial component of the high-severity area plant community, comprising nearly half of the mean total cover. Shrub dominance was also reflected in the identification of three shrub species as indicators of high severity as noted above (Table 3). Cover did not differ significantly by treatment for any of these life forms, although the means generally followed the trends in total plant cover (Table 2 and Fig. 3).

Cover of exotic forbs was significantly higher ($p = 0.004$) in high-severity burned areas (0.68%) versus low-severity areas (0.11%) (Fig. 3 and Table 2). This difference was driven primarily by two species, *Tragopogon dubius* Scop. (yellow salsify) and *Verbascum thapsus* L. (common mullein), that were commonly found on high-severity plots (32 of 35 plots and 30 of 35 plots, respectively). Although their overall cover values were low, the prevalence of these species is reflected in their designation as an indicator species of high-severity areas (Table 3). There was not a significant treatment effect for exotic forbs. No significant patterns were detected for exotic graminoids for either severity or treatment (Table 2).

### 3.4. Pine regeneration

Pine regeneration density was highly variable, and so we present these data but conducted statistical tests on differences in fre-

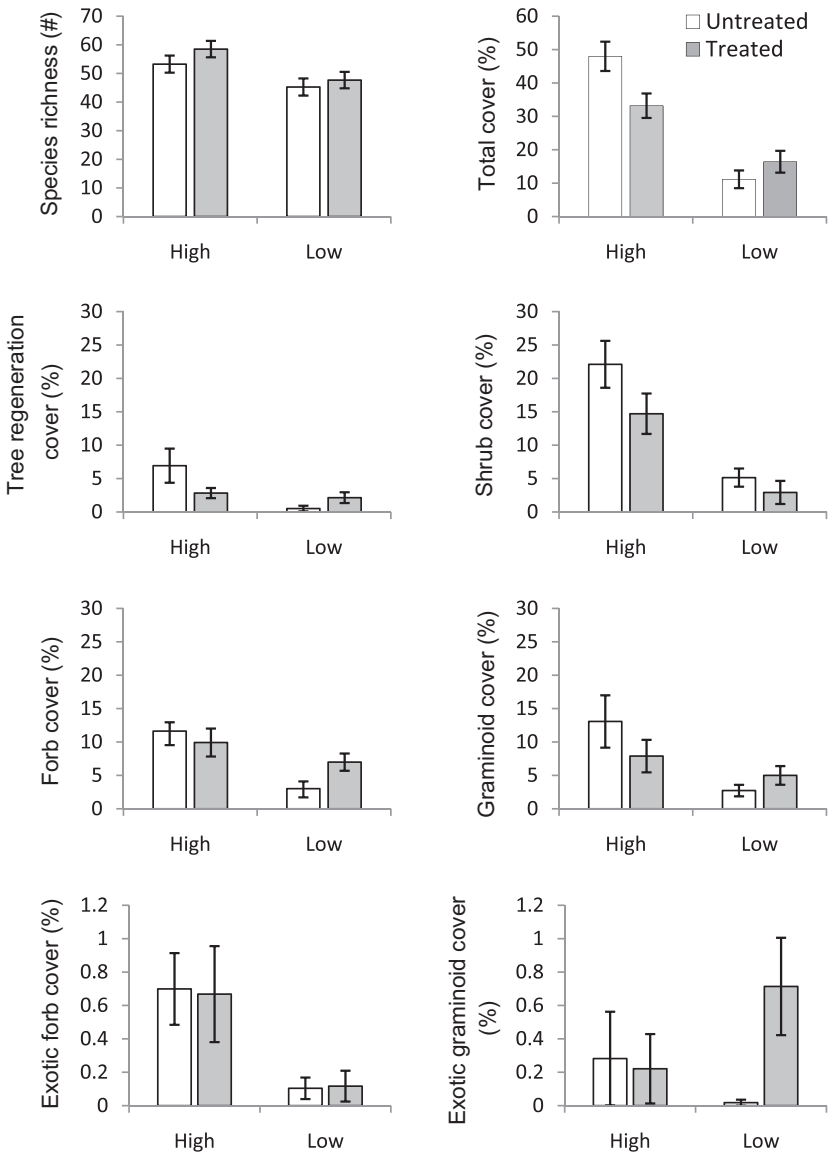


**Fig. 3.** Species richness, total plant canopy cover, and cover by life form by severity class and pre-fire treatment, with standard error. Note differences in scale on the *y*-axis.

quency. Pine regeneration relative frequency for seedlings in the "established" 41–137 cm height class were significantly more frequent ($p = 0.006$) in high-severity areas (0.51) compared to low-severity areas (0.31). In high severity areas, seedling density in the 41–137 cm height class averaged 888.9 seedlings ha$^{-1}$ (median 329.5) in high-severity areas and 68.64 seedlings ha$^{-1}$ (median 0) in low-severity areas. For the combined height classes, seedling density ranged from 0 to 4283.2 seedlings ha$^{-1}$ in high severity areas and from 0 to 659.0 seedlings ha$^{-1}$ in low-severity areas. Pine regeneration relative frequency for seedlings in the "established" 41–137 cm height class was also significantly different by treatment ($p = 0.008$), where the frequency was 0.43 in treated areas versus 0.11 in untreated areas.

When examining pine regeneration within low-severity areas, this treatment effect resulted in a mean pine regeneration frequency of 0.17 in treated areas and 0.06 in untreated areas (Tables 2 and 4). Pine seedling density in this size class averaged 123.6 (median 0) seedlings ha$^{-1}$ in low-severity treated areas and 13.7 (median 0) in low-severity untreated areas. When the height classes were combined, seedling densities ranged from 82.4 to 659.0 seedlings ha$^{-1}$ on low-severity treated sites versus 0–329.47 on low-severity untreated sites. In high-severity areas, mean pine seedling frequency in the 41–137 cm height class was 0.67 in treated areas and 0.19 in untreated areas. Pine regeneration density in this size class averaged 1345.4 seedlings ha$^{-1}$ (median 1276.7) in high-severity treated areas and 432.44 (median 0) in high-severity untreated areas (Table 4). When all height classes were combined, seedling densities on high-severity treated sites ranged from 164.7 to 4283.2 seedlings ha$^{-1}$ versus 0 to 2347.5 seedlings ha$^{-1}$ on high-severity untreated sites.

### 3.5. Neighborhood severity

Finally, the examination of neighborhood severity surrounding high-severity sites detected an average of 47.2% low- and moderate-severity areas combined for the pre-fire untreated sites. This was significantly lower ($p = 0.035$) than the 73.1% combined low- and moderate-severity in the neighborhood of the pre-fire treated sites (Fig. 4).

## 4. Discussion

Fire severity had a persistent impact on post-fire vegetation characteristics 8 years post-fire. We observed large and significant

differences between low- and high-severity burned areas for almost all variables measured. In high-severity areas, the resources made available after the loss of mature trees likely enabled the higher mean understory plant canopy cover, of which nearly half was comprised of shrubs. This pattern is consistent with that observed in the earlier investigation by Kuenzi et al. (2008). The importance of shrubby species in high-severity areas is underscored by the appearance of three shrubs (*A. pringlei, A. pungens, C. fendleri*) and a re-sprouting tree that exhibits shrub-like patterns of juvenile growth (*R. neomexicana*) as indicator species, none of which were indicators of high-severity in the earlier investigations. The combination of few to no overstory trees and high shrub cover has resulted in shrub-dominated communities, unlike low-severity areas where mature ponderosa pine dominance was maintained post-fire.

We detected low cover of exotic species throughout the project area (<2%) in 2010, which is only slightly lower than that reported in the initial investigation (<3%) (Kuenzi et al., 2008). Drought conditions for the period of recovery may be contributing to the limited exotic response, and low propagule pressure may also play a role (Lockwood et al., 2005). The low exotic response detected on this fire is consistent with that observed by Foxx (1996) and Huisinga et al. (2005), but it is inconsistent with several studies that detected significant invasions in severely burned southwestern ponderosa pine forests. However, it is difficult to compare our data with these studies because they either did not explicitly state which exotic species were observed (Griffis et al., 2001) or they used different definitions of species' nativity (Barclay et al., 2004; Crawford et al., 2001).

Despite the overall low exotic plant cover on the Rodeo–Chediski Fire, we did observe some important differences by severity class. In contrast to earlier measurements, exotic forb cover was significantly higher in high-severity burned areas, although it was still quite low at <1%. The minute difference in exotic forb cover by severity may not be very biologically meaningful, but the increases in frequency since the 2004/2005 measurements for some exotic species on high-severity plots lend some credence to a difference between severities. This difference was driven by the prevalence of two exotic forbs, *T. dubius* and *V. thapsus*. Although cover values for each were relatively low, the number of plots with these species steadily increased since the initial investigation in 2004, particularly for *T. dubius*. This species was found on 51.4% more high-severity plots in 2010 than in 2004, and *V. thapsus* was found on 11.4% more high-severity plots. This pattern is further reflected in the designation of both species as indicator species for high-severity in 2010, which was not the case in 2004 or 2005. The increase of *V. thapsus* through time in cover and frequency among plots was also observed on the Apache-Sitgreaves National Forest portion of this fire (Shive et al., 2013), but is in contrast to the reductions through time detected elsewhere on the Rodeo–Chediski Fire (Ffolliott et al., 2011), and after restoration treatments elsewhere in the Southwest (Stoddard et al., 2011). On our sites, the increasing prevalence of these species may become cause for concern because propagule buildup could potentially facilitate more significant invasions in the future, given the availability of suitable habitat (Lockwood et al., 2005; Moles et al., 2012).

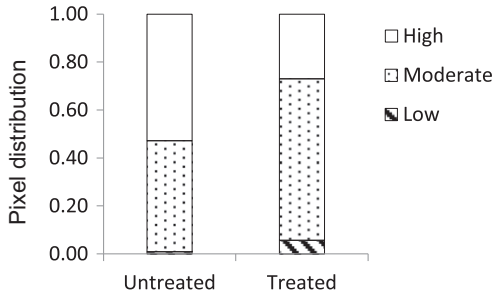


**Fig. 4.** Neighborhood severity distribution for high severity treated and untreated areas.

More generally, we observed slightly lower exotic species richness than that reported in the initial investigation by Kuenzi et al. (2008), and also detected some changes in exotic species diversity and abundance. Of the exotic species found on at least 5% of the plots, five species observed in 2004 and 2005 were not included on the 5% subset list in 2010: *Agropyron desertorum* (Fisch. ex Link) Schult. (desert wheatgrass), *Dactylis gomerata* L. (orchardgrass), *T. aestivum*, *Cirsium vulgare* (Savi) Ten. (bull thistle), and *Chenopodium album* L. (lambsquarters). Our lack of detection of *C. album* may not represent an actual change, since the *Chenopodium* plants we encountered were in a juvenile stage that did not allow us to distinguish between native and exotic *Chenopodium* species. The common wheat seeded immediately post-fire was frequent in 2004, infrequent in 2005 (Kuenzi et al., 2008), and entirely absent in 2010. Our data show that the annual cereal grain did quickly exit the system as intended by managers (Peppin et al., 2010). Four exotic species that were observed infrequently in 2004/2005 were detected on at least 5% of plots in 2010: *E. curvula*, *Poa compressa* L. (Canada bluegrass), *P. pratensis*, and *Erodium cicutarium* (L.) L'Hér. ex Aiton (redstem stork's bill). The increase in frequency of these species through time may also be cause for concern in the future if the trend continues. Only the perennial forb, *Sanguisorba minor* Scop. (small burnet), was completely new in 2010, yet it occurred on only two plots.

It is important to note that some of these exotic species may have been introduced to the area via past seeding practices. The perennial grass *E. curvula* was known to have been seeded in the course of past management activities before the Rodeo–Chediski Fire (Kuenzi et al., 2008), and many of the other exotic species we found have been seeded as part of management activities throughout the Southwest: *P. pratensis*, *Melilotus officinalis* (L.) Lam. (sweetclover), *D. glomerata* (Fowler et al., 2008). In addition to past intentional seeding of exotic species, native seed mixes (such as those used after the Rodeo–Chediski Fire) have the potential for exotic species contamination (Fowler et al., 2008; Keeley, 2006; Peppin et al., 2010). Because of this risk of contamination, even seeding with sterile species such as common wheat, which did exit the system as intended, should be applied with caution.

**Table 4**
Pine regeneration mean relative frequency and density (standard errors in parentheses) by severity and treatment, in the 0–40 cm and 41–137 cm height classes, as well as the two classes combined ("All seedlings").

| Severity/treatment | Relative frequency | | | Density (seedlings ha$^{-1}$) | | |
|---|---|---|---|---|---|---|
| | 0–40 cm | 41–137 cm | All <137 cm | 0–40 cm | 41–137 cm | All <137 cm |
| *Low* | | | | | | |
| Untreated | 0.17(0.11) | 0.06(0.06) | 0.17(0.14) | 82.4(56.3) | 13.7(13.7) | 96.1(61.7) |
| Treated | 0.39(0.06) | 0.17(0.11) | 0.44(0.07) | 164.7(56.3) | 123.6(84.4) | 288.3(107.9) |
| *High* | | | | | | |
| Untreated | 0.06(0.06) | 0.19(0.12) | 0.25(0.12) | 13.7(13.7) | 432.4(385.1) | 446.2(382.3) |
| Treated | 0.28(0.10) | 0.67(0.09) | 0.78(0.07) | 288.3(209.2) | 1345.4(435.7) | 1633.6(617.6) |

Pre-fire treatment had no effect on exotic response, but did have significant impacts on several response variables. The detection of significant differences in pine regeneration, total plant cover, and plant community composition by pre-fire treatment suggests that although severity was the predominant driver of many vegetation differences, pre-fire treatments can influence post-fire communities. In particular, treatment effects on pine regeneration and total understory plant cover are likely to have important implications for post-fire reforestation. In the low-severity areas, the higher mean frequency in treated areas may be important for concerns about seedling recruitment in post-settlement forests. When all seedling height classes are combined, each of the pre-fire treated sites had regeneration on at least one plot. In contrast, four of the six untreated low-severity sites completely lacked ponderosa pine regeneration on all three plots.

The pine seedling densities we detected are difficult to compare directly to other studies in areas with intact forest structure due to differences in the definition of seedling size. These definitions varied from recruits less than breast height (Keyser et al., 2008), to those $\geqslant$20 cm in height and <2.54 cm DBH $*$(Puhlick et al., 2012), to those $\leqslant$5 cm DBH (Bailey and Covington, 2002). We detected an average of 96 seedlings ha$^{-1}$ in low-severity untreated plots and 288 seedlings ha$^{-1}$ in low-severity treated plots; whereas Bailey and Covington (2002) detected much lower seedling density (18–60 seedlings ha$^{-1}$) despite their more inclusive definition of a seedling. However, all of these observations are considerably lower than observed by Keyser et al. (2008) (up to 2000 seedlings ha$^{-1}$), and Lentile et al. (2005) (612 seedlings ha$^{-1}$) in the more mesic Black Hills of South Dakota. They are also lower than what Puhlick et al. (2012) reported in the southwestern US (536–14,184 seedlings ha$^{-1}$). Although significant mortality is possible (Smith et al., 1997), the densities we observed in the low-severity treated areas (288 seedlings ha$^{-1}$) may be adequate to provide the 0.4–3.6 trees ha$^{-1}$, per decade that Mast et al. (1999) determined recruited into the overstory under the pre-European fire regime.

In high-severity areas, the treatment effect on pine regeneration frequency may be important for reforestation of severely burned areas. Given the unknown future patterns of climate and disturbance, a myriad of trajectories are possible on these landscapes. However, the current frequencies of pine regeneration suggest that untreated areas in high-severity burned sites are more likely to remain shrub-dominated; whereas treated sites within high-severity areas have a better chance of recruitment back to pine-dominated forest. In untreated areas, the low frequency and density of pine seedlings within high-severity sites indicate a higher likelihood of conversion to shrublands. In the larger 41–137 cm size class, only two of six sites had any ponderosa pine regeneration. For both sizes combined, regeneration occurred on only three of six sites, averaging 446 seedlings ha$^{-1}$ for both size classes, and two of these sites had densities <250 seedlings ha$^{-1}$. This number is slightly higher than those reported in several other studies in the ponderosa pine type. On the La Mesa Fire in northern New Mexico, Foxx (1996) reported roughly 81–121 seedlings ha$^{-1}$; whereas in high-severity areas of the 2000 Jasper Fire in the Black Hills, no seedlings were detected in severely burned areas in either year three post-fire (Lentile et al., 2005) or in year five post-fire (Keyser et al., 2008). The densities at these sites are far below the $\sim$809 seedlings ha$^{-1}$ Pearson (1950) recommended to fully re-stock clear-cut areas under even-aged management (although Pearson does not specify size class in this recommendation).

We attribute this limited pine recruitment in untreated areas within high-severity sites in part to distance to seed source. In these areas, pre-fire treatment functioned to limit high-severity burn patches, and thereby create more heterogeneous areas surrounding our sampling sites. There were more low- and moderate-severity burned areas in the neighborhood of sites with pre-fire treatments where high-severity patch size was limited; whereas untreated high-severity sites were predominantly surrounded by more high-severity burned area (i.e. situated in larger patches of high severity). This likely explains the treatment effect for relative frequency of pine regeneration in high-severity areas, because low- and moderate-severity areas generally maintain more overstory structure that can provide a seed source.

Other studies in ponderosa pine have shown decreased pine recruitment with increasing distance from unburned, or lightly burned, edge (Bonnet et al., 2005; Haire and McGarigal, 2008, 2010; Lentile et al., 2005). We also observed lower rates of recruitment in areas farther from lesser-burned edges (in our study, the proxy for this was the low and moderate severity areas that were less prevalent around the untreated sites). By increasing the size of high-severity burned patches, these recent "mega-fires" are increasing the distance to edge/seed source, which is impacting forest regeneration. We recognize that regeneration patterns are complex, and that all large high-severity patches may not face the same limitations. Long-distance seed dispersal and surviving, isolated overstory trees within a high-severity patch may act to reduce the effective patch size in terms of distance to seed source.

In addition to limited seed source availability, site capture by shrubs or grasses may also inhibit future pine regeneration. Several studies in severely burned ponderosa pine forests have linked increased shrub cover to decreased pine regeneration (Haire and McGarigal, 2008, 2010; Strom and Fulé, 2007). We similarly observed significantly lower pine regeneration on untreated sites, where we also observed significantly higher total plant canopy cover. This cover was primarily comprised of graminoids, shrubs, and re-sprouting tree species that exhibit shrub-like growth forms in their juvenile stage (such as *Quercus gambelii* Nutt. (Gambel oak), *Juniperus deppeana* Steud. (alligator juniper), and *R. neomexicana*).

In contrast, the pre-fire treated sites within high-severity had significantly more frequent pine regeneration compared to untreated sites. This suggests that these areas generally have a better chance at reforestation than the untreated sites. However, two of the treated sites had excessively high pine seedling densities, which may indicate a trajectory towards overly dense stands. The densities at these sites (>2000 seedlings ha$^{-1}$) greatly exceed the $\sim$809 seedlings ha$^{-1}$ Pearson (1950) recommended to fully re-stock clear-cut areas. Similarly high densities of pine regeneration in severely burned areas have been reported in longer-term investigations by Savage and Mast (2005) and Haire and McGarigal (2010). Without management intervention, these high-density sites may be at risk of re-burning severely in the future.

## 5. Management recommendations

The results from this project fit into a body of literature that does not indicate a straightforward or easily predictable pattern of exotic response, suggesting more intensive investigations are needed. Where existing plots burn over, re-measurements should be a priority in order to better understand how pre-fire species assemblages interact with severity and management treatments to influence post-fire plant communities. In addition, the results of this investigation support the continued application of management practices that reduce fuels, such as uneven-aged silvicultural strategies or thinning and burning treatments. These treatments are important to the persistence of ponderosa pine forests because they generally reduce fire severity where applied, and even where burning conditions permit high-severity fire in treated areas, our neighborhood severity analysis suggests that the patch size is likely to be smaller than in untreated areas, which may significantly impact post-fire pine regeneration. Finally, the detection

of important differences in vegetation response by pre-fire treatment within severity class demonstrates that landscapes that burned under a given severity class are not necessarily alike. As such, managing post-fire landscapes under the assumptions that all low- or high-severity areas are the same may ignore important differences in plant communities and future trajectories.

## Acknowledgements

We thank the White Mountain Apache Tribe for access and support for the project, particularly Tribal Forester Jonathan Brooks. Funding was provided by the Joint Fire Science Program (JFSP 11-1-1-27), the Ecological Restoration Institute (ERI), and the National Fire Plan (FWE-3) through Challenge Cost Share Agreement 09-CS-190 with the Rocky Mountain Research Station (RMRS).

## References

Anderson, M.J., 2001. Permutation tests for univariate or multivariate analysis of variance and regression. Can. J. Fish. Aquat. Sci. 58, 626–639.
Avery, T.E., Burkhart, H.E., 2001. Forest Measurements, fifth ed. McGraw-Hill, New York.
Bailey, J.D., Covington, W.W., 2002. Evaluating ponderosa pine regeneration rates following ecological restoration treatments in northern Arizona, USA. For. Ecol. Manage. 155, 271–278.
Barclay, A.D., Betancourt, J.L., Allen, C.D., 2004. Effects of seeding ryegrass (*Lolium multiflorum*) on vegetation recovery following fire in a ponderosa pine (*Pinus ponderosa*) forest. Int. J. Wildland Fire 13, 183–194.
Baron, J.S., Gunderson, L., Allen, C.D., Fleishman, E., McKenzie, D., Meyerson, L.A., Oropeza, J., Stephenson, N., 2009. Options for national parks and reserves for adapting to climate change. Environ. Manage. 44, 1033–1042.
Barton, A.M., 2002. Intense wildfire in southeastern Arizona: transformation of a Madrean oak-pine forest to oak woodland. For. Ecol. Manage. 165, 205–212.
Bonnet, V.H., Schoettle, A.W., Shepperd, W.D., 2005. Postfire environmental conditions influence the spatial pattern of regeneration for *Pinus ponderosa*. Can. J. For. Res. 35, 37–47.
Brooks, M.L., D'Antonio, C.M., Richardson, D.M., Grace, J.B., Keeley, J.E., DiTomaso, J.M., Hobbs, R.J., Pellant, M., Pyke, D., 2004. Effects of invasive alien plants on fire regimes. Bioscience 54, 677–688.
Byrne, R., McAndrews, J.H., 1975. Pre-Columbian Purslane (*Portulaca-Oleracea* L) in New World. Nature 253, 726–727.
Cochrane, M.A., Morgan, C.J., Wimberly, M.C., Baer, A.D., Finney, M.A., Beckendorf, K.L., Eidenshink, J., Zhu, Z., 2012. Estimation of wildfire size and risk changes due to fuel treatments. Int. J. Wildland Fire, <http://dx.doi.org/10.1071/WF11079>.
Covington, W.W., Moore, M.M., 1994. Southwestern ponderosa forest structure - changes since Euro-American settlement. J. Forest. 92, 39–47.
Crawford, J.S., Wahren, C.H.A., Kyle, S., Moir, W.H., 2001. Responses of exotic plant species to fires in *Pinus ponderosa* forests in northern Arizona. J. Veg. Sci. 12, 261–268.
Daubenmire, R.F., 1959. A canopy-coverage method. Northwest Sci. 33, 43–64.
Dodge, R.S., Fulé, P.Z., Sieg, C.H., 2008. Dalmatian toadflax (*Linaria dalmatica*) response to wildfire in a southwestern USA forest. Ecoscience 15, 213–222.
Dufrene, M., Legendre, P., 1997. Species assemblages and indicator species: The need for a flexible asymmetrical approach. Ecol. Monogr. 67, 345–366.
Ellison, A.M., Bank, M.S., Clinton, B.D., Colburn, E.A., Elliott, K., Ford, C.R., Foster, D.R., Kloeppel, B.D., Knoepp, J.D., Lovett, G.M., Mohan, J., Orwig, D.A., Rodenhouse, N.L., Sobczak, W.V., Stinson, K.A., Stone, J.K., Swan, C.M., Thompson, J., Von Holle, B., Webster, J.R., 2005. Loss of foundation species: consequences for the structure and dynamics of forested ecosystems. Front. Ecol. Environ. 3, 479–486.
Ffolliott, P.F., Stropki, C.L., Chen, H., Neary, D.G., 2011. The 2002 Rodeo–Chediski Wildfire's Impacts on Southwestern Ponderosa Pine Ecosystems, Hydrology and Fuels. USDA For. Serv. Res. Paper RMRS-RP-85.
Finney, M.A., McHugh, C.W., Grenfell, I.C., 2005. Stand- and landscape-level effects of prescribed burning on two Arizona wildfires. Can. J. For. Res. 35, 1714–1722.
Fowler, J.F., Sieg, C.H., Dickson, B.G., Saab, V., 2008. Exotic plant species diversity: influence of roads and prescribed fire in Arizona ponderosa pine forests. Rangeland Ecol. Manage. 61, 284–293.
Foxx, T.S., 1996. Vegetation succession after the La Mesa fire at Bandelier National Monument. In: Allen, C.D. (Ed.), Fire Effects in Southwestern Forests: Proceedings of the Second La Mesa Fire Symposium, 29–31 March, 1994.* USDA Forest Service, Rocky Mountain Forest and Range Experiment Station, Fort Collins, CO, Los Alamos, NM, pp. 47–69.
Fulé, P.Z., Covington, W.W., 1998. Spatial patterns of Mexican pine-oak forests under different recent fire regimes. Plant Ecol. 134, 197–209.
Fulé, P.Z., Crouse, J.E., Roccaforte, J.P., Kalies, E.L., 2012. Do thinning and/or burning treatments in western USA ponderosa or Jeffrey pine-dominated forests help restore natural fire behavior? For. Ecol. Manage. 269, 68–81.
Griffis, K.L., Crawford, J.A., Wagner, M.R., Moir, W.H., 2001. Understory response to management treatments in northern Arizona ponderosa pine forests. For. Ecol. Manage. 146, 239–245.
Haire, S.L., McGarigal, K., 2008. Inhabitants of landscape scars: succession of woody plants after large, severe forest fires in Arizona and New Mexico. Southwest. Nat. 53, 146–161.
Haire, S.L., McGarigal, K., 2010. Effects of landscape patterns of fire severity on regenerating ponderosa pine forests (*Pinus ponderosa*) in New Mexico and Arizona, USA. Landsc. Ecol. 25, 1055–1069.
Huisinga, K.D., Laughlin, D.C., Fulé, P.Z., Springer, J.D., McGlone, C.M., 2005. Effects of an intense prescribed fire on understory vegetation in a mixed conifer forest. J. Torrey Bot. Soc. 132, 590–601.
Institute of Ecosystem Studies, 1999. Gap Light Analyzer (Computer Program). Simon Fraser University. <http://www.ecostudies.org/gla>.
JMP, Version 7. SAS Institute Inc. Cary, NC, 1989–2007.
Keeley, J.E., 2006. Fire management impacts on invasive plants in the western United States. Conserv. Biol. 20, 375–384.
Keyser, T.L., Lentile, L.B., Smith, F.W., Shepperd, W.D., 2008. Changes in forest structure after a large, mixed-severity wildfire in ponderosa pine forests of the Black Hills, South Dakota, USA. For. Sci. 54, 328–338.
Kuenzi, A.M., Fulé, P.Z., Sieg, C.H., 2008. Effects of fire severity and pre-fire stand treatment on plant community recovery after a large wildfire. For. Ecol. Manage. 255, 855–865.
Larson, M.M., Schubert, G.H., 1970. Cone crops of ponderosa pine in central Arizona including the influence of Albert squirrels. USDA For. Serv. Res. Paper RM-58.
Lentile, L.B., Smith, F.W., Shepperd, W.D., 2005. Patch structure, fire-scar formation, and tree regeneration in a large mixed-severity fire in the South Dakota Black Hills, USA. Can. J. For. Res. 35, 2875–2885.
Lockwood, J.L., Cassey, P., Blackburn, T., 2005. The role of propagule pressure in explaining species invasions. Trends Ecol. Evol. 20, 223–228.
Mast, J.N., Fulé, P.Z., Moore, M.M., Covington, W.W., Waltz, A.E.M., 1999. Restoration of presettlement age structure of an Arizona ponderosa pine forest. Ecol. Appl. 9, 228–239.
McCune, B., Grace, J.B., 2002. Analysis of Ecological Communities. MjM Software Design, Gleneden Beach, Oregon.
McCune, B., Mefford, M.J., 1999. PC-ORD (Computer Software). Multivariate Analysis of Ecological Data, Version 4. MjM Software Design, Gleneden Beach, OR.
McDonald, P.M., 1980. Seed dissemination in small clear cuttings in north-central California. USDA For. Serv. Res. Paper PSW-150, 5.
Moles, A.T., Flores-Moreno, H., Bonser, S.P., Warton, D.I., Helm, A., Warman, L., Eldridge, D.J., Jurado, E., Hemmings, F.A., Reich, P.B., Cavender-Bares, J., Seabloom, E.W., Mayfield, M.M., Sheil, D., Djietror, J.C., Peri, P.L., Enrico, L., Cabido, M.R., Setterfield, S.A., Lehmann, C.E.R., Thomson, F.J., 2012. Invasions: the trail behind, the path ahead, and a test of a disturbing idea. J. Ecol. 100, 116–127.
Pearson, G.A., 1950. Management of Ponderosa Pine the Southwest as developed by research and experimental practice. USDA For. Serv. Agricultural Monograph 6, 217, Washington, DC.
Peck, J.E., 2010. Multivariate Analysis for Community Ecologists: Step-by-Step using PC-ORD. MjM Software Design, Gleneden Beach, OR.
Peppin, D., Fule, P.Z., Sieg, C.H., Beyers, J.L., Hunter, M.E., 2010. Post-wildfire seeding in forests of the western United States: an evidence-based review. For. Ecol. Manage 260, 573–586.
Pollet, J., Omi, P.N., 2002. Effect of thinning and prescribed burning on crown fire severity in ponderosa pine forests. Int. J. Wildland Fire 11, 1–10.
Puhlick, J.J., Laughlin, D.C., Moore, M.M., 2012. Factors influencing ponderosa pine regeneration in the southwestern USA. For. Ecol. Manage 264, 10–19.
Savage, M., Mast, J.N., 2005. How resilient are southwestern ponderosa pine forests after crown fires? Can. J. For. Res. 35, 967–977.
Seager, R., Ting, M.F., Held, I., Kushnir, Y., Lu, J., Vecchi, G., Huang, H.P., Harnik, N., Leetmaa, A., Lau, N.C., Li, C.H., Velez, J., Naik, N., 2007. Model projections of an imminent transition to a more arid climate in southwestern North America. Science 316, 1181–1184.
Shive, K.L., A.M. Kuenzi, C.H. Sieg, P.Z. Fule, 2013. Pre-fire fuel reduction treatments influence plant communities and exotic species 9 years after wildfire. Applied Veg. Science. http://dx.doi.org/10.1111/avsc.12015.
Smith, D.M., Larson, B.C., Kelty, M.J., Ashton, P.M.S., 1997. The Practice of Silviculture: Applied Forest Ecology. John Wiley & Sons, New York.
Stoddard, M.T., McGlone, C.M., Fulé, P.Z., Laughlin, D.C., Daniels, M.L., 2011. Native plants dominate understory vegetation following ponderosa pine forest restoration treatments. West. N. Am. Nat. 71, 206–214.
Strom, B.A., 2005. Pre-fire treatment effects and post-fire forest dynamics on the Rodeo–Chediski burn area, Arizona. M.S. Thesis. School of Forestry, Northern Arizona University, Flagstaff AZ.
Strom, B.A., Fulé, P.Z., 2007. Pre-wildfire fuel treatments affect long-term ponderosa pine forest dynamics. Int. J. Wildland Fire 16, 128–138.
USDA-NRCS, 2006. The PLANTS Database. National Plant Data Center, Baton Rouge, LA 70874-4490, USA, 29 November, 2006. <http://plants.usda.gov>.
USDA-NRCS, 2011. The PLANTS Database. National Plant Data Center, Baton Rouge, LA 70874-4490, USA, 15 November, 2011. <http://plants.usda.gov>.
Welsh, S.L., Atwood, N.D., Goodrich, S., Higgins, L.C., 1993. A Utah Flora. Brigham Young University, Provo, UT.
Westerling, A.L., Hidalgo, H.G., Cayan, D.R., Swetnam, T.W., 2006. Warming and earlier spring increase western US forest wildfire activity. Science 313, 940–943.