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Effects of fire severity and pre-fire stand treatment on plant community recovery after a large wildfire

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Abstract

The Rodeo–Chediski fire burned approximately 189,650 ha in east–central Arizona from June 18 to July 7, 2002, 113,700 ha of it on White Mountain Apache tribal land. In 2004 and 2005, we measured plant canopy cover and richness in areas of high and low burn severity in each of two treatments: (1) cutting and prescribed burning, or (2) untreated, in the 11 years prior to the wildfire. Total understory plant canopy cover was significantly higher in areas of high severity ($p = .0002$ in 2004 and $p = .0001$ in 2005). Overall, there was high richness of exotic species but cover was surprisingly low at <3% across all years, severities, and treatments. There were no significant differences in exotic species cover between high and low severity or between treated and untreated areas. Areas of high severity burn were seeded after the fire with several native grasses, native forbs, and common wheat (*Triticum aestivum* L.). Wheat had a strong presence in the plant community in 2004, but was uncommon by 2005. Indicator Species Analysis showed the indicators of high severity were seeded or early successional species. In contrast, indicators of low severity included several perennial bunchgrasses. While our results show that wheat declined quickly and exotic plants were uncommon, any longer-term impacts of the fire on the plant community can only be addressed by continued monitoring of these sites.
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Keywords: Non-native species; Southwest; Ponderosa pine; Plant community; Wheat

1. Introduction

Increasing numbers of large wildfires in western U.S. forests in recent decades (Westerling et al., 2006) have raised concerns about long-term ecological consequences. In addition to the loss of timber and increased soil erosion, large wildfires may greatly alter post-fire plant communities by providing ideal habitat for a number of exotic species (Keeley et al., 2003; Crawford et al., 2001). When existing plant biomass is removed by a disturbance resources become available to invading exotic plants (Bataineh et al., 2006; Swope, 2003). Exotic plant invasions threaten the ecological health of forests world-wide and can alter fire regimes (Brooks et al., 2004; Lake and Leishman, 2004).

Grime (1977) outlined three strategies for plant establishment in response to stress and disturbance, the ruderal (R-selection), competitive (C-selection), and stress-tolerant (S-selection) strategies of plant evolution. Ruderal species are adapted to environments with high disturbance but low stress, and therefore colonize areas that are high in nutrients and other resources after disturbance. These species are often considered ‘‘early successional’’ species and are commonly annual or biennial herbs (D’Antonio and Chambers, 2006; Lake and Leishman, 2004).

In southwestern ponderosa pine (*Pinus ponderosa* P. & C. Lawson) forests, Griffis et al. (2001) compared understory response between stands that were thinned, thinned and prescribed burned, burned by stand-replacing wildfire, and unmanaged control stands. They found that while overall plant canopy cover increased with treatment intensity (the most intense being stand-replacing wildfire, the least intense being thinned only), exotic species cover and species richness increased dramatically after wildfire. Native graminoid cover and species richness were higher on forested stands compared to areas severely burned in the 1996 wildfire. Crawford et al. (2001) noted that higher species richness after moderate to high severity fire was due to exotic species and native ruderal species such as *Conyza canadensis* (L.) Cronq. (Canadian

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horseweed). In contrast, a study by Huisinga et al. (2005) found that native plants were the key taxa that increased total plant species richness and abundance after a high intensity fire in Grand Canyon National Park. They attributed the limited exotic species response to the fact that this area was never logged and rarely grazed by livestock (grazing in GCNP ceased in the mid 1930s), so there were few or no exotic species seeds present in the seed bank. Other factors, such as the remoteness of their study area, lack of nearby roads, and the Park policy of not seeding after fire, may have contributed to the paucity of exotic species. A 16-year study by Foxx (1996) on the 1977 La Mesa fire in Bandelier National Monument, NM, also found few exotic species, perhaps because of the ''preservation'' management strategy of the National Park Service. One seeded species, slender wheatgrass (*Elymus trachycaulus* (Link) Gould ex Shinners), was initially a dominant species after the fire but dropped out within a decade. Native ruderal species, such as fetid goosefoot (*Chenopodium graveolens* Willd.), were also dominant right after the fire and eventually disappeared.

The Rodeo–Chediski fire is the largest and most severe wildfire on record in the Southwest. It burned approximately 189,650 ha in east–central Arizona from June 18 to July 7, 2002, leaving a mosaic pattern of burn severity on 113,700 ha of White Mountain Apache tribal land. The fire burned through several vegetation types including chaparral, piñon-juniper woodland (*Pinus edulis* Engelm.-*Juniperus* spp.), ponderosa pine forest, and isolated pockets of mixed conifer (Finney et al., 2005; Strom, 2005; USDA Forest Service, 2002). Many forested stands within the fire's perimeter had been managed by the tribe for decades, utilizing commercial timber harvests, non-commercial thinning, and prescribed burning. The severity of the Rodeo–Chediski fire was substantially reduced in areas that had received recent (<11 years pre-fire) management treatments (Finney et al., 2005; Strom, 2005). Management treatments that reduce fuels and thus the severity of subsequent wildfire can have multi-century impacts.

We investigated the effect of the pre-fire forest management and fire severity on plant community recovery after the Rodeo–Chediski fire. Given the timber management on the White Mountain Apache tribal land and the varied severity patterns created by the Rodeo–Chediski fire, a matrix of conditions was present. We hypothesized that fire severity would override pre-fire timber management practices in affecting the post-fire plant community. That is, severely burned areas would resemble each other whether or not pre-fire treatments had occurred. We hypothesized that high burn severity areas would have: (1) higher total plant canopy cover; (2) greater representation of ruderal and seeded species as indicators; and (3) higher exotic species richness and plant canopy cover, compared to areas classified as low burn severity, based on the plant colonization strategies outlined by Grime (1977). We also expected that species seeded post-burn would attain higher cover in high severity areas, and that *Triticum aestivum* L. (common wheat), seeded post-fire, would be a prominent species in seeded areas, but would not persist over time.

2. Methods

2.1. Site selection

The White Mountain Apache tribal lands lie south of the Mogollon Rim in central-eastern Arizona. Maximum and minimum temperatures for the region, as reported at the Heber Ranger Station by the Western Regional Climate Center (www.wrcc.dri.edu) range from 29.3 °C in July to −8.8 °C in January, based on 1950–2005 averages. Average total annual precipitation is 44.4 cm (standard deviation 11.5 cm). Total annual precipitation during the years of data collection was lower than the 55-year average: 33.3 cm in 2004 and 40.7 cm in 2005. Total snowfall in winter of 2003/2004 was 50.3 and 22.9 cm in winter of 2004/2005, compared to the long term average of 97.3 cm. Elevation for the study area ranges from 2000 to 2295 m. All sites were above 2000 m to ensure they fell in a ponderosa pine-dominated ecosystem, and at less than 45% slope. The average slope was 17.2%. Study sites were randomly selected within the combinations of two levels of fire severity (low and high) and two types of pre-fire forest management practices (cut and burned and untreated; henceforth referred to as treatments). Fire severity was determined from a remotely sensed Differenced Normalized Burn Ratio (ΔNBR) map. This map shows the difference between the infrared and near infrared wavelengths emitted by pre-fire and post-fire vegetation ($\Delta NBR = NBR_{pre\text{-}fire} - NBR_{post\text{-}fire}$). High severity areas show a greater difference than low severity areas due to greater vegetation die-off. The thresholds for high and low severity were based on Cocke et al. (2005), who classified burn severity in a nearby national forest. Low severity included unburned areas and areas that had a ΔNBR of less than 570; high severity areas had a ΔNBR greater than 570. The treatment categories were 'cut and burned' within 11 years prior to the fire or 'no treatment' within the same period. The 11-year time period was determined by Strom's (2005) analysis of treatment boundaries compared with burn severity; forest areas treated >11 years were indistinguishable in the severity of burn from untreated areas. The boundaries of these treatments were provided by the tribe. For consistency, plots were restricted to soil types of the Overgaard series including Overgaard gravelly fine sandy loam and Overgaard gravelly loam. This was the most common soil type found within the fire perimeter. These are well-drained alluvial soils derived from sandstone, quartzite, and granite (Mitchell, 1981) and are classified as fine, mixed, active, frigid Typic Paleustalfs (Soil Survey Staff, 2006).

High severity areas of the Rodeo–Chediski fire were seeded during Burned Area Emergency Rehabilitation (BAER) in 2002 after the fire. The seed mix used on White Mountain Apache tribal lands included the exotic species common wheat, applied at a rate of 16.8 kg ha^{-1}. Common wheat is often seeded in order to create a non-persistent ground cover to control erosion and noxious weed invasion (Keeley, 2004). The remainder of the seed mix included: *E. trachycaulus* (3.8 kg ha^{-1}), western wheatgrass (*Pascopyrum smithii* (Rydb.) A. Löve) (2.8 kg ha^{-1}), switchgrass (*Panicum virgatum* L.) (0.8 kg ha^{-1}), green needlegrass (*Nassella viridula* (Trin.) Barkworth) (1.6 kg ha^{-1}), mountain brome (*Bromus marinatus* = *Bromus carinatus* H. & A.) (Welsh et al., 1993) (2.7 kg ha^{-1}), sideoats grama (*Bouteloua curtipendula* (Michx.) Torr.) (0.4 kg ha^{-1}), sand dropseed (*Sporobolus cryptandrus* (Torr.) Gray) (0.005), plains coreopsis (*Coreopsis tinctoria* Nutt.) (0.07 kg ha^{-1}), purple prairie clover (*Dalea purpurea* Vent.) (0.04 kg ha^{-1}), blue flax (*Linum lewisii* Pursh) (0.06 kg ha^{-1}), and black-eyed susan (*Rudbeckia hirta* L.) (0.03 kg ha^{-1}) (J. A. Youtz, pers. comm., 2003).

2.2. Measurement

We used a stratified random design with two levels of pre-fire treatments (treated/untreated) and two levels of fire severity (low/high). Six study sites, with three plots each, situated 200 m apart, were measured in each severity and treatment combination. Plot centers were a subset of those used by Strom (2005) to assess overstory tree structure after the wildfire. At each plot center, we established two 44.8-m transects, crossing perpendicularly, resulting in a 31.7 m × 31.7 m square, approximately 1000 m^2. We measured plant canopy cover in 20 cm × 50 cm quadrats spaced at 4-m intervals along each transect, for a total of 20 quadrats for each plot. Within each quadrat, we measured total plant canopy cover, cover of forbs, exotic forbs, graminoids, exotic graminoids, shrubs, and trees <1.4 m tall, plus cover by species using six cover classes (Daubenmire, 1959). We also estimated cover of litter, rocks, logs, bare ground, and moss. Plant nomenclature and classification of species as native or exotic follows USDA-NRCS (2006) and Welsh et al. (1993); one exception was *Portulaca oleracea* L. (little hogweed), which was listed as ‘introduced’ on the PLANTS database, but we classified as ‘native’ based on Byrne and McAndrews’ (1975) evidence that it was present in North America before the late 1400s. Voucher specimens are stored at the White Mountain Apache Tribe’s office in Whiteriver, Arizona. We grouped species to the generic level when vegetative characteristics were insufficient to identify to the species level. In addition, we quantified plant species richness for the entire plot by recording all species that occurred within each 1000-m^2 plot. Overstory canopy cover was estimated from a hemispherical photograph at each plot center using a digital camera with a 180° fisheye lens (Nikon CoolPix E4300 and FC-E8 Fisheye Converter Lens). Hemispherical photos were analyzed using Gap Light Analyzer (Institute of Ecosystem Studies, 1999) to quantify percent canopy openness. Overstory data and physical characteristics (aspect, slope) were measured in 2004. Pre-fire basal area was calculated by adding the basal areas of surviving and fire-killed trees. Our measurements took place from mid-July to early September in 2004 and 2005, in order to coincide with monsoonal rain.

2.3. Statistical methods

We analyzed the data using DISTLM (Anderson, 2001), a non-parametric, permutation procedure for analysis of variance for multivariate data. We used DISTLM to test for multivariate differences between plant communities in high and low severity areas to test our hypothesis that fire severity would override pre-fire timber management practices in the post-fire plant community. We also used DISTLM to test our hypotheses that high burn severity areas would have higher total plant canopy cover and higher exotic species richness and plant canopy cover, compared to areas classified as low burn severity. DISTLM was used for analyzing univariate measures for richness and abundance because these data evidenced non-normality and heterogeneous variances. We carried out 9999 permutations for each test. Euclidean distance was used for univariate data and the Bray-Curtis dissimilarity measure was used for plant community data. The alpha level was 0.05.

We performed non-metric multi-dimensional scaling ordination of the cover data by species using PC-ORD software (McCune and Mefford, 1999). This was done as a qualitative measure in order to provide a visual display of the data and the factors driving the differences between the plant communities. The ordination was done using the Bray-Curtis distance measure (Faith et al., 1987). The parameters were set such that 40 runs were performed with real data, 50 runs were performed with randomizations (to determine how likely the observed stress value of the final solution would be by chance alone), a maximum of 400 iterations per run, and an instability criterion of 0.00001. Finally, we used PC-ORD to identify Indicator Species of treatments and fire severity classes. Indicator species values were calculated for each species as relative frequency × percent canopy cover. Species with $p < .05$ and an indicator value >25 (Dufrêne and Legendre, 1997) were identified as indicator species. The threshold value of 25 was arbitrarily set by Dufrêne and Legendre, supposing that an indicator species is present in at least 50% of the sites in one group (in this case the treatment and fire severity classes) and that its percent canopy cover in that group reaches at least 50%. The Indicator Species Analysis allowed us to see if species

Table 1
Overstory characteristics of sites on White Mountain Apache Tribal lands by pre-fire treatments and severities

Severity	Treatment	Canopy openness post-fire (%)	Density (trees/ha) post-fire	Density (trees/ha) pre-fire	Basal area (m^2/ha) post-fire	Basal area (m^2/ha) pre-fire
High	Untreated	66.9 (2.3)	44.2 (35.9)	508.5 (137.5)	1.9 (1.5)	15.1 (3.1)
	Cut and burned	65.1 (1.6)	9.9 (4.6)	632.5 (168.7)	0.9 (0.4)	12.0 (1.7)
Low	Untreated	39.4 (2.4)	647.5 (147.6)	937.6 (184.1)	21.6 (2.8)	25.5 (3.0)
	Cut and burned	45.6 (2.1)	343.0 (65.3)	728.7 (222.6)	14.0 (1.8)	17.8 (2.8)

Pre-fire data are based on standing live and dead trees. Post-fire data are based on live trees only. Data are means (standard error).

seeded post-burn were characteristic of high severity areas, including *Triticum aestivum* L. (common wheat).

3. Results

The pre-fire forest was composed mainly of ponderosa pine, but also included Douglas-fir (*Pseudotsuga menziesii* (Mirbel) Franco), Gambel oak (*Quercus gambelii* Nutt.), white fir (*Abies concolor* (Gord. and Glend.) Lindl. ex Hildebr.), and junipers (*Juniperus osteosperma* (Torr.) Little and *Juniperus deppeana* Steud.). There was no significant difference in pre-fire basal area between treatment types ($p = .563$) or between fire severities ($p = .136$). After the fire, in high severity areas across both treatments, live tree density declined by 95% and basal area of live trees declined by 90% (Table 1). In low severity areas live tree density declined by 40% and basal area of live trees declined by only 18%. Post-fire high severity areas had no living *A. concolor* or *P. menziesii* trees. Basal area of live trees was significantly lower in high severity areas compared to low severity areas ($p = .039$) after the fire. There was no significant difference for treatment effect ($p = .781$). Canopy openness post-burn was significantly higher ($p = .019$) in high severity areas (>65%) than in low severity areas (<46%).

We found a total of 310 plant species across all sites and years, but we focused on the 151 species found on at least 5% of the plots and for analysis of cover we used only the 87 species found within at least 5% of the transect quadrats. Removal of species that occur at frequencies of <5% is often recommended to avoid having the analysis driven by very uncommon species (Gauch, 1982). Total plant species richness was not significantly different for any combination of severity and treatment in either year (Table 2, Fig. 1a). Analyzing differences in richness for all forbs and all graminoid species separately, there were no differences in forb richness for severity or treatment but graminoids were significantly richer in high severity burns in 2005 and in areas that had been cut and burned in 2004 (Table 2).

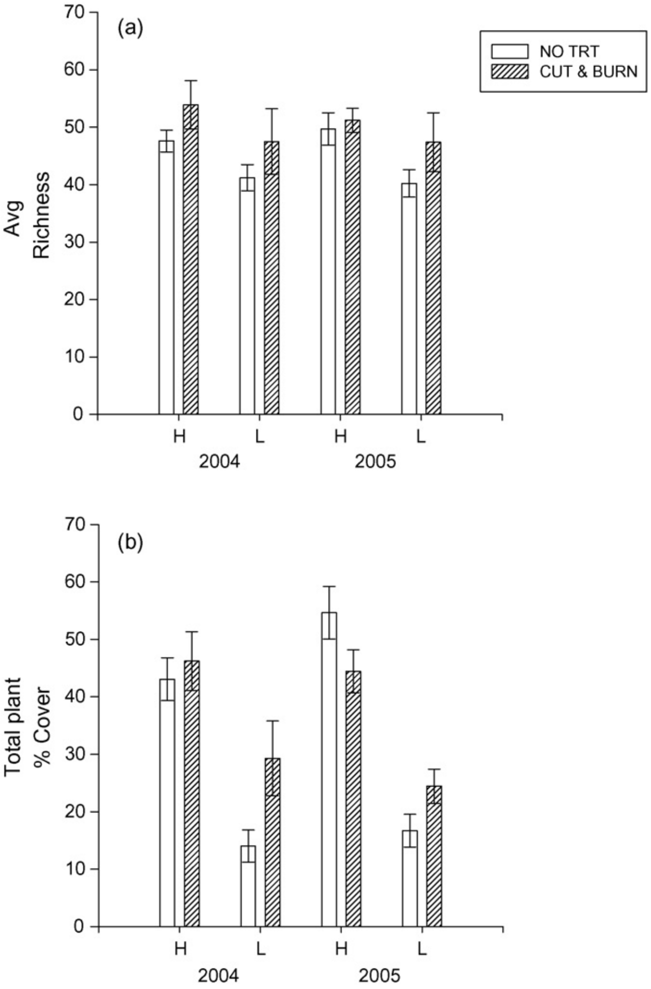

Fig. 1. (a) Average (+S.E.) richness on 1000 m^2 plot by fire severity (H = high, L = low), treatment, and year. No significant difference was found for treatment or severity in either year. (b) Average (+S.E.) canopy cover on 20- by 50-cm quadrats by fire severity (H = high, L = low), treatment, and year. In 2004, a significant difference was found for the severity effect ($p = .0002$). The interaction between severity and treatment was significant ($p = .0233$) in 2005.

Table 2

A summary of the permuted p-values for DISTLM tests for univariate plant canopy cover and richness, by category

Cover

	Severity		Treatment		Interaction	
Category	2004	2005	2004	2005	2004	2005
Cover						
Total cover	0.0002*	0.0001*	0.0675	0.7403	0.2165	0.0233*
Forbs	0.0069*	0.0002*	0.0908	0.5402	0.5921	0.2846
Exotic Forbs	0.2202	0.0619	0.7133	0.5139	0.5785	0.6411
Graminoids	0.0137*	0.0014*	0.0737	0.9288	0.9144	0.5401
Exotic Graminoids	0.0695	0.9705	0.7419	0.7570	0.4874	0.2176
Shrubs	0.0251*	0.0086*	0.6120	0.5163	0.6663	0.3300
Trees	0.0049*	0.0831	0.1714	0.3117	0.0020*	0.0006*
Richness						
Total richness	0.1115	0.0565	0.1181	0.2037	0.9857	0.4153
Forbs	0.4098	0.2662	0.0999	0.0960	0.7640	0.4610
Graminoids	0.0876	0.0471*	0.0421*	0.1531	0.8841	0.9034

[*] Indicates significance ($p < .05$).

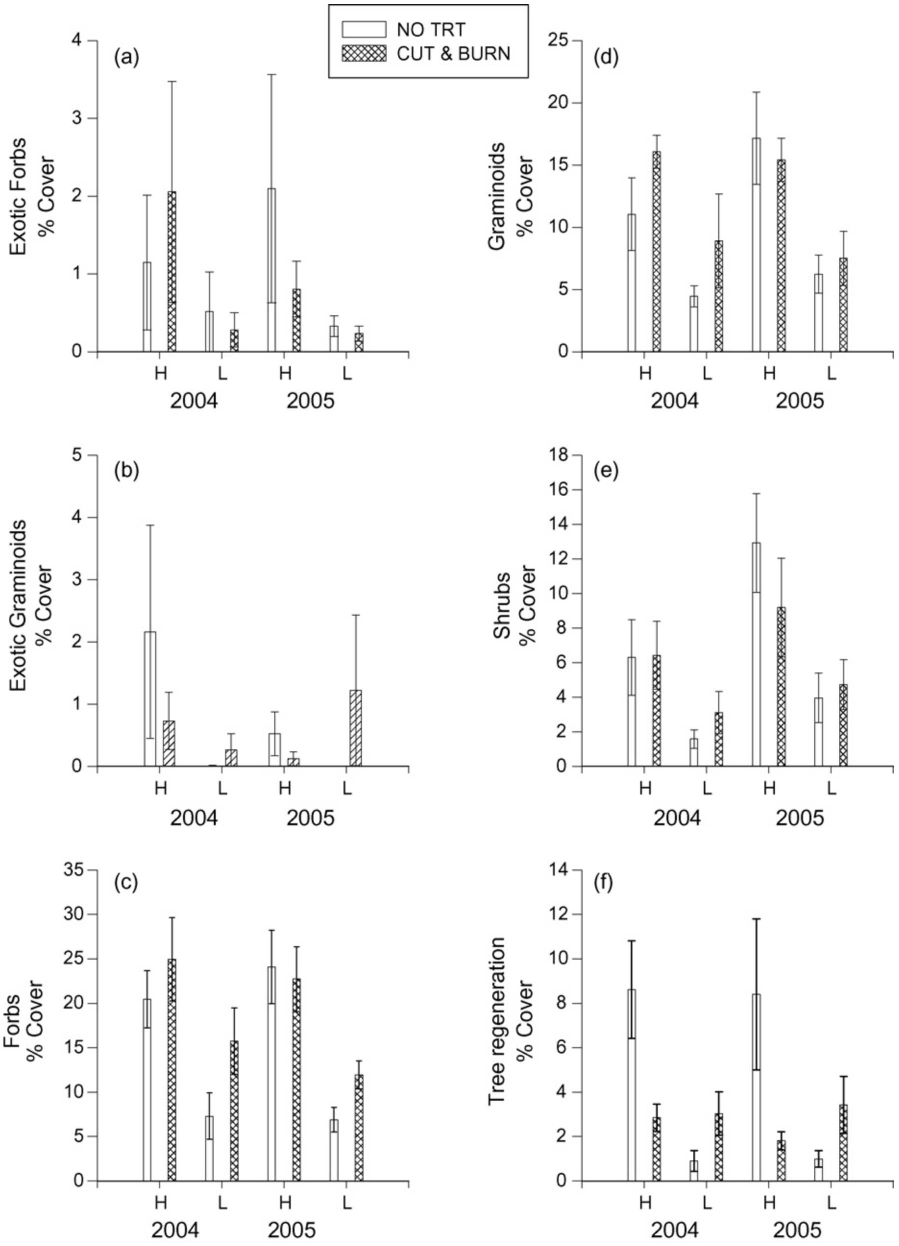

Fig. 2. Average (+S.E.) cover, by plant groups, fire severity (H = high, L = low), treatment, and year. (a) Exotic forbs: no significant differences. (b) Exotic graminoids: no significant differences. (c) All forbs: a significant difference was found between high and low severity in 2004 at $p = .007$ and in 2005 at $p = .0002$. (d) All graminoids: a significant difference was found between high and low severity in 2004 at $p = .0137$ and in 2005 at $p = .0014$. (e) Shrubs: a significant difference was found between high and low severity in 2004 at $p = .025$ and in 2005 at $p = .009$. (f) Tree regeneration (<1.4 m tall): the interaction between fire severity and treatment was significant in 2004 at $p = .0137$ and in 2005 at $p = .0014$.

Total plant cover was significantly higher in high severity areas than on low severity areas in 2004, supporting our hypothesis that fire severity was more influential than pre-wildfire forest management practices (Table 2, Fig. 1). However, the interaction between severity and treatment was significant for total plant cover in 2005. Contrary to our hypothesis that high severity areas would have more exotic cover than low severity areas, neither exotic forbs nor exotic graminoids showed significant differences for severity or treatment effect (Fig. 2a and b). There were 35 exotic species overall, but 13 were found on at least 5% of the plots and were included in analyses (Table 3). Total forb, graminoid, and shrub cover was significantly higher in high severity areas but showed no significant difference for treatment effects, for both 2004 and 2005 (Fig. 2c–e). Tree regeneration cover had a significant interaction between severity and treatment effects in both 2004 and 2005, indicating that the response to fire severity was not consistent between treatments (Fig. 2f).

Multivariate assessment of plant canopy cover of individual species showed significant differences for severity effect ($p = .0001$ in both years). Plant community composition based on cover data by species was significantly different for treatment effect in 2004 ($p = .044$), but not in 2005 ($p = .10$). This appeared to be driven by one species, *Robinia*

Table 3
Summary of observed exotic species by fire severity

Severity	Species present[a]		
	Seeded	Noxious	Other
High	***Agropyron desertorum***	***Bromus tectorum***	***Bromus japonicas***
	Bromus inermis[b]	***Cirsium vulgare***	***Chenopodium album***
	Dactylis glomerata	*Convolvulus arvensis*	*Echinochloa crus-galli*
	Eragrostis curvula	*Erodium cicutarium*	*Kochia scoparia*
	Lolium perenne	*Onopordum acanthium*	***Lactuca serriola***
	Triticum aestivum	***Verbascum thapsus***	*Lolium arundinaceum*
			Malva neglecta
			Medicago lupulina
			Melilotus officinalis[b]
			Poa compressa[b]
			Poa pratensis
			Polygonum aviculare
			Polygonum convolvulus[b]
			Polypogon viridis
			Rumex acetosella
			Setaria viridis
			Sonchus asper
			Taraxacum officinale
			Tragopogon dubius
Low	***Agropyron desertorum***	***Bromus tectorum***	***Bromus japonicas***
	Bromus inermis	***Cirsium vulgare***	***Chenopodium album***
	Dactylis glomerata	*Convolvulus arvensis*	*Eragrostis lehmanniana*
	Eragrostis curvula	***Verbascum thapsus***	***Lactuca serriola***
	Triticum aestivum		*Lolium arundinaceum*[b]
			Medicago lupulina
			Melilotus officinalis[b]
			Poa compressa[b]
			Poa pratensis
			Polygonum aviculare
			Polygonum convolvulus[b]
			Rumex acetosella
			Rumex crispus
			Taraxacum officinale
			Tragopogon dubius

Classification of ‘noxious’ is based on designation in at least one of four southwestern states (Sieg et al., 2003). Species in the ‘seeded’ category may have been seeded throughout the known history of the White Mountain Apache tribal lands, not uniquely as part of the BAER efforts (pers. comm., J. A. Youtz, 2006). A total of 31 exotic species were found on high severity plots and 24 on low severity plots.

[a] Species that are found on >5% of the plots within the entire study (not by severity) are in bold. These species were used in all analyses.

[b] Species found only in 2005.

neomexicana Gray (New Mexico locust), which had nearly 32 times more cover in untreated areas than in areas that were cut and burned. Our analysis of composition of the plant community based on presence/absence of species indicated significant differences for both treatment (2004, $p = .0036$; 2005, $p = .0046$) and severity effects ($p = .0001$ both years). In each case, the difference was driven by very few species.

Ordinated plant communities were separated by severity-level based on plant canopy cover data, providing a qualitative illustration of the differences in plant community cover data (Fig. 3a and b). However, when grouped by treatments instead of by severities the distinction could not be made, supporting the results of the permutation procedure. No separation was evident using presence/absence data either, although a significant difference was found using the permutation procedure.

We identified 9 indicator species for high severity in 2004 and 10 in 2005 (Table 4). *B. carinatus*, *E. trachycaulus*, and common wheat were seeded species. *Chenopodium album* L. (lambsquarters), *Chenopodium graveolens*, *Cirsium wheeleri* (Gray) Petrak (Wheeler’s thistle), and *Conyza canadensis* were indicator species in both 2004 and 2005 on high severity sites. In 2005, additional high severity indicator species were *Bahia dissecta* (Gray) Britt. (ragleaf bahia), *B. curtipendula*, *P. smithii*, and *Pseudognaphalium macounii* (Greene) Kartesz (Macoun’s cudweed). In low severity areas, *Koeleria macrantha* (Ledeb.) J. A. Schultes (prairie Junegrass), *Muhlenbergia longiligula* A. S. Hitchc. (longtongue muhly), *Packera neomexicana* (Gray) W. A. Weber and A. Löve (New Mexico groundsel), and *Poa fendleriana* (Steud.) Vasey (muttongrass) were indicator species in 2004. *K. macrantha* and *P. neomexicana* dropped out as indicators of low severity sites in 2005.

Frequency of common wheat dramatically declined from 2004 to 2005 (Fig. 4). In 2004, wheat was highest in high severity areas that had received no pre-fire treatment. In low

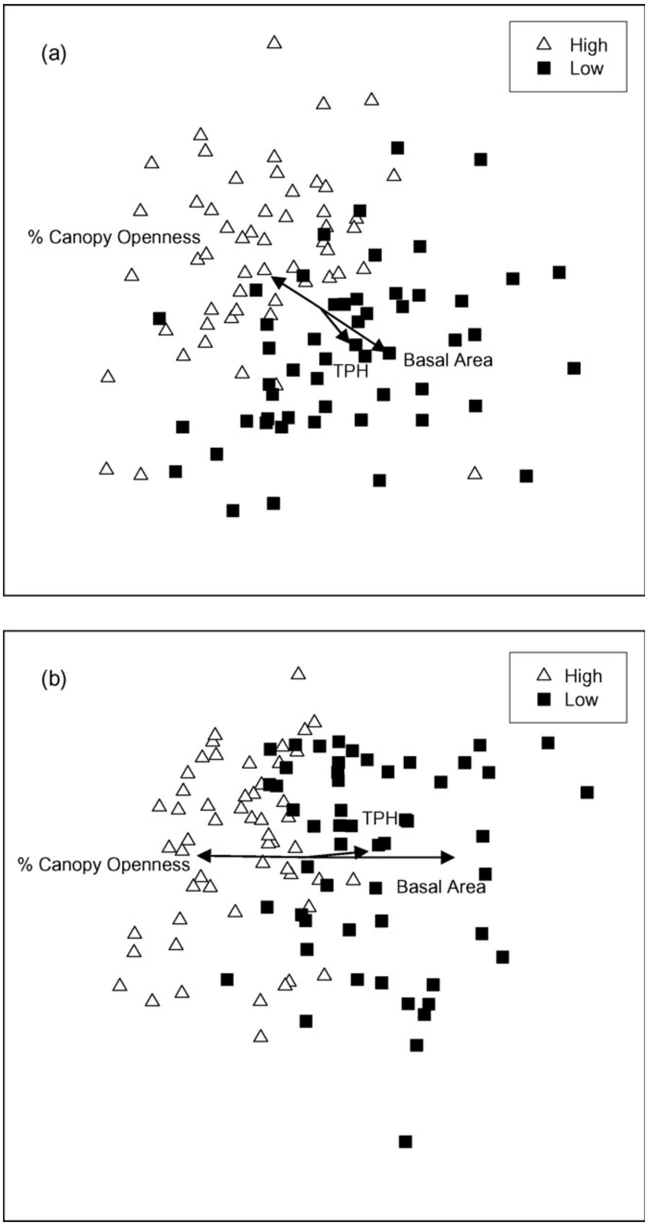

Fig. 3. Ordinated plant communities based on plant cover data by severity-level. Vector length indicates strength of correlation. (a) 2004 data. Stress level is 19.71 for a 3-d solution. Low severity plots were positively correlated with basal area and trees per hectare. (b) 2005 data. Stress level is 19.31 for a 3-d solution. Low severity plots were also positively correlated with basal area and trees per hectare.

severity areas in 2004, wheat was only detected in areas that had not received treatment, and was not detected at all on low severity sites in 2005.

4. Discussion

Our comparison of areas of high and low burn severity within areas that had been previously cut and burned (treated) and untreated areas provided support for the hypothesis that fire severity would play a more important role in the post-wildfire plant community than treatment. There was significantly higher plant canopy cover in 2004 in high severity sites compared to low severity burned sites, but no difference due to treatment effect. Sabo (2006) found that basal area must be reduced to $8\ \mathrm{m}^2\ \mathrm{ha}^{-1}$ or less to result in an increase in understory production. Clary and Ffolliott (1996) found an understory response at 5–18 $\mathrm{m}^2\ \mathrm{ha}^{-1}$. Basal area in the high severity sites after the fire was $1.9\ \mathrm{m}^2\ \mathrm{ha}^{-1}$ in untreated sites and $0.9\ \mathrm{m}^2\ \mathrm{ha}^{-1}$ in treated sites, clearly low enough to result in an understory response. In low severity areas basal area was 21.6 and $14.0\ \mathrm{m}^2\ \mathrm{ha}^{-1}$, respectively.

We predicted that there would be greater cover of exotic plants in areas that burned with higher severity; to our surprise, this hypothesis was not upheld. Other studies in national parks and monuments, which are managed for preservation, have shown relatively low cover of exotics. Several studies in more actively managed landscapes, including national forests, have shown higher exotic cover (however, see Fornwalt et al., 2003). This study and the accompanying research we did on the Apache-Sitgreaves NF have shown low exotic cover despite active management and fire use over a long period of time (Table 5). Huisinga et al. (2005) found 28% total plant cover 6 years after an intense prescribed fire, and of that, $<1\%$ was exotic plant cover. Eight years post-burn, they found 50% total plant cover, and only 1.0% was exotic species. They attributed the near absence of exotic plants in their study to the fact that the area had not been heavily managed or grazed. In contrast, the lands of the White Mountain Apache reservation have been managed for decades for timber and grazing. Therefore, there is reason to suspect that propagule pressure of exotic species would be sufficient for populations of a number of previously introduced exotic species to expand following the fire, but the exotic species cover is similar with $<1\%$ cover in both 2004 and 2005.

Our results contrast with those of Crawford et al. (2001), who found twice as much exotic cover on high severity burns as on moderate burns within 2 years of the wildfire. Some of the difference between our study and Crawford's was due to different authorities for whether a species was exotic. Crawford et al. (2001) classified *Conyza canadensis* and *Chenopodium graveolens* as exotics. We classify both species as native based on USDA-NRCS (2006). The exotic species we found on at least 5% of plots, and therefore included in analysis, included only four designated as noxious in one or more southwestern state, *Bromus tectorum* L. (cheatgrass), *Cirsium vulgare* (Savi) Ten. (bull thistle), *Erodium cicutarium* (L.) L'Hér. Ex. Ait. (redstem storksbill) and *Verbascum thapsus* L. (common mullein). Other noxious species, *Onopordum acanthium* L. (Scotch thistle) and *Convolvulus arvensis* L. (field bindweed) were found on fewer than 5% of plots. Noticeably absent were other noxious species adapted to high fire severity conditions: *Linaria dalmatica* (L.) P. Mill. ssp. *dalmatica* (Dalmatian toadflax) (Dodge et al., in press), *Salsola tragus* L. (Russian thistle) (Crawford et al., 2001), and *Centaurea diffusa* (Wolfson et al. (2005). Most of the exotic species we found are not considered noxious in southwestern states, were found in very low cover, or were seeded (Table 3, Fig. 2a and b).

Uncertainties about plant identification and whether plant species are native or exotic can make an assessment of the role

Table 4
Indicator species by year and level of fire severity

Year	Severity	Species	Indicator value	*p*-Value	Average cover (S.E.)	Comments[a]
2004	High	*Bromus carinatus*	72.2	0.0010	4.8 (0.8)	Seeded species
		Ceanothus fendleri	61.3	0.0020	13.3 (1.7)	Fire resprouter
		Chenopodium album[b]	37.4	0.0010	1.0 (0.4)	Disturbance adapted
		Chenopodium graveolens	63.9	0.0100	1.9 (0.4)	Disturbance adapted
		Cirsium wheeleri	33.5	0.0230	2.0 (0.5)	Disturbance adapted
		Conyza canadensis	67.1	0.0010	2.2 (0.4)	Disturbance adapted
		Elymus trachycaulus	59.1	0.0010	3.9 (0.5)	Seeded species
		Lotus wrightii	61.1	0.0010	3.4 (0.5)	
		Triticum aestivum[b]	42.5	0.0010	1.6 (0.6)	Seeded species
	Low	*Koeleria macrantha*	32.1	0.0170	0.9 (0.2)	
		Muhlenbergia longiligula	35.3	0.0030	3.4 (0.9)	
		Packera neomexicana	47.9	0.0010	0.2 (0.04)	
		Poa fendleriana	40.2	0.0010	0.8 (0.2)	
2005	High	*Bahia dissecta*	68.7	0.0010	3.6 (0.7)	Disturbance adapted
		Bouteloua curtipendula	31.0	0.0040	1.4 (0.4)	Seeded species
		Bromus carinatus	68.0	0.0010	3.9 (0.6)	Seeded species
		Ceanothus fendleri	58.7	0.0030	18.5 (2.2)	Fire resprouter
		Cirsium wheeleri	48.3	0.0020	3.1 (0.8)	Disturbance adapted
		Conyza canadensis	68.6	0.0010	2.0 (0.4)	Disturbance adapted
		Elymus trachycaulus	63.9	0.0010	3.9 (0.6)	Seeded species
		Lotus wrightii	52.6	0.0100	3.1 (0.5)	
		Pascopyrum smithii	41.8	0.0010	2.1 (0.6)	Seeded species
		Pseudognaphalium macounii	38.7	0.0440	0.9 (0.3)	Disturbance adapted
	Low	*Muhlenbergia longiligula*	37.4	0.0020	4.7 (1.1)	
		Poa fendleriana	26.9	0.0270	0.7 (.2)	

[a] ‘‘Disturbance adapted’’ indicates that species has been associated with burned areas in other studies. ‘‘Seeded species’’ indicates this species was in the seed mix used on the burned area.

[b] Non-native species.

of wildfires in enhancing exotic species confusing. Using different authorities can lead to different conclusions about whether or not a species is classified as exotic. Similar to Crawford et al. (2001), Barclay et al. (2004) classified *Conyza canadensis* as an exotic species, which comprised most of the total exotic cover on their sites (Table 5). Had they classified *C. canadensis* as a native species, their assessment of the abundance of exotic species would have been different. Some papers do not give their source for whether a species is classified as native or exotic, and others do not provide lists of species classified as exotic. Researchers can help clarify some of these issues by collecting voucher specimens, identifying their source used to identify exotic species, and providing a list of species classified as exotic and their contribution to total exotic abundance.

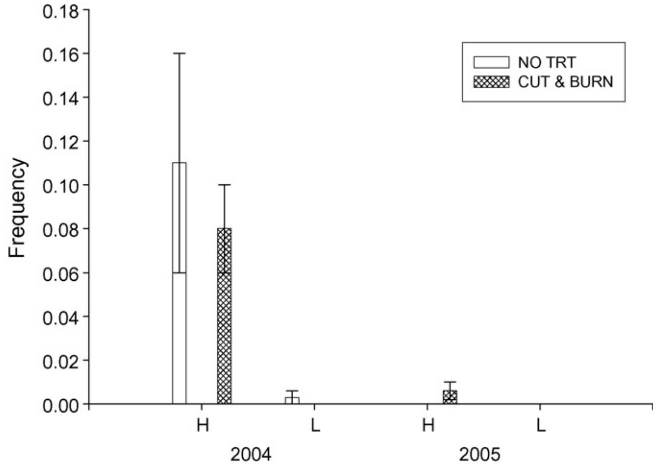

Fig. 4. Average frequency (proportion of quadrats + S.E.) of *Triticum aestivum*, or common wheat, by year, fire severity (H = high, L = low) and treatment.

It might have been logical to expect that some exotic plant species would establish following disturbances due to seed bank germination. While we had no pre-fire vegetation data, seeding had been done in past decades as part of rehabilitation after logging operations on skid trails, log landings, and roads. Exotic species were used for seeding until the late 1990s, when the policy was revised to using only native species. In the mid 1990s the White Springs fire was revegetated using a seed mix that included *Agropyron desertorum* (Fisch. ex Link) J. A. Schultes (desert wheatgrass), *Dactylis glomerata* L. (orchard-grass), and *Bromus inermis* Leyss. (smooth brome). *Eragrostis curvula* (Schrad.) Nees (weeping lovegrass) was also seeded on road closures on the eastern side of the fire, but did not spread across the landscape (J. A. Youtz, personal communication, 2006). While these species were detected in our plots, they were minor species.

Precipitation and droughts greatly affect understory plant community composition and productivity in the southwest (Milchunas, 2006; Moore et al., 2006). The study area received more total precipitation in 2005 than it did in 2004, but there was over twice as much snowfall in 2004. These levels of

Table 5
A comparison of cover and richness of exotic plant species following severe wildfires in the Southwest

Site author (year)	Time since fire	Method for quantifying cover	# of plots	Cover of exotics[a] (area sampled/plot)	Richness of exotics[a] (total area sampled)	Notes
White Mtn. Apache Tribal land, AZ Kuenzi (2006)	2 years, 3 years	Cover class	71	0.9% in 2004, 0.7% in 2005 ($2\ m^2 = 20 \times .1\ m^2$)	35, 13 on >5% plots[b] (≈71,000 m^2)	This study.
Apache-Sitgreaves NF, AZ Kuenzi (2006)	2 years, 3 years	Cover class	84	0.3% in 2004, 1.6% in 2005 ($2\ m^2 = 20 \times .1\ m^2$)	38, 14 on >5% plots[b] (≈84,000 m^2)	Also Rodeo–Chediski fire
Kaibab and Coconino NF, AZ Crawford et al. (2001)	2 years	Cover class	100	116% in high sev[c] ($2.4\ m^2 = 24 \times 0.1\ m^2$)	30r[c] (40,000 m^2)	Data were pooled from three wildfires
Coconino NF, AZ Griffis et al. (2001)	<5 years	11 min search and abundance rating	10	≈50% forbs, ≈8% graminoids (375 m^2)	39[d] (60,000 m^2)	Data based only on the 10 wildfire plots
Grand Canyon NP, AZ Huisinga et al. (2005)	6 years, 8 years	Point intercept	60	0.7% in 1999, 1% in 2001 (2, 50-m transects, 166 pts ea.)	4 (60,000 m^2)	This was an intense prescribed fire
Bandelier Nat'l Monument, NM	2 years	Line intercept	49	≈18% in high sev[e]	?	Data based only on non-seeded plots in 1998
Barclay et al. (2004)				≈15% in mod sev[e] (50-m transect)		
Bandelier Nat'l Monument, NM Foxx (1996)	8 years	Cover class	?	≈0.1% (50 quadrats)	3 (1858 m^2)	Data based only on the first 8 years of data collection after the fire

[a] Tallied for the entire study, not per plot or per m^2.

[b] All analysis was done on species occurring on over 5% of the plots.

[c] Calculated by summing average cover values of individual species; includes five species which *USDA-NRCS (2006) categorizes as native, plus Amarathus sp., (<0.5%), which includes species classified as both native and exotic*.

[d] The species were not listed and therefore we were uncertain if the same species were classified 'exotic' as in this study.

[e] *C. canadensis* was classified as an exotic and included in this measure.

precipitation were less than the 55-year averages. Perhaps insufficient precipitation, especially snowfall, played a role in limiting the response of annual species, including exotics.

When we looked at the plant community data by species composition, we found the species with significantly higher cover in high severity sites matched closely with the results from the Indicator Species Analysis. Of the 13 species found to be indicator species for high severity (Table 4), 5 were specifically seeded in high severity areas (*Bouteloua curtipendula*, *Bromus carinatus*, *E. trachycaulus*, *P. smithii* and common wheat). Whether these seeded native grass species will remain dominant over time remains to be seen. Foxx (1996) observed that the seeded grass, *E. trachycaulus*, was dominant in the years after the 1977 La Mesa fire, but was not observed after 1985. The remaining eight species are disturbance adapted, or increase in response to fire. *Ceanothus fendleri* Gray (Fendler's ceanothus) resprouts vigorously after fire, and dormant seeds in the seed bank are stimulated to germinate by heat (Huffman and Moore, 2004). Both *Chenopodium album* and *Chenopodium graveolens* are disturbance adapted (Laughlin et al., 2004), and Crawford et al. (2001) found *C. album* to have a strong response to high severity burn. *B. dissecta*, *Cirsium wheeleri*, *Conyza canadensis*, and *P. macounii* are disturbance tolerant or dependent on disturbances for their establishment, following the Grime (1977) model of ruderal species establishment. *Lotus wrightii* (Gray) Greene (Wright's deervetch) was shown to be a dominant species on the La Mesa fire by Foxx (1996) and DiTomaso et al. (1999) report *Lotus* spp. increase in response to fire.

Four species were indicators of low severity sites. *K. macrantha*, *M. longiligula*, and *P. fendleriana* are all perennial bunch grasses, which are adapted to low severity fire by sprouting from leaf bases insulated by old leaf sheaths (Bond and van Wilgen, 1996) or from below the soil surface where they are protected (Young, 1983). In addition, *R. neomexicana* was an indicator of low severity in 1 year.

The significant differences in plant species composition based on presence or absence of species for treatment and severity effects were driven by few species in each case. Of the 155 species found on at least 5% of the plots, 14 were unique to untreated areas, 8 unique to cut and burned, 8 unique to high severity areas, and 4 to low severity areas. These small differences can probably be accounted for more by the pre-existing plant community than it can to actual wildfire or treatment effects. An exception may be some of the species that occurred in high severity but not on low severity in 2005. These included *Amaranthus hybridus* L. (slim amaranth), *Ambrosia acanthicarpa* Hook. (flatspine burr ragweed), and *P. oleracea* which are known to grow in disturbed areas (Ecological Restoration Institute, 2005).

Our hypothesis that wheat would not persist over time was upheld. Wheat nearly disappeared from our plots by 2005. Keeley (2004) suggested that the large die-off of wheat could create "an ecological vacuum," and therefore present opportunity for invasion of exotics. While our research can not fully address this concern, we observed a minor, but non-significant increase in exotic forb cover in 2005 in only one category, untreated/high severity, which increased from 1.15% cover (S.E. = 0.87) to 2.1% cover (S.E. = 1.47) in 2005. All other combinations of severity and treatment showed slight declines in exotic forb cover in 2005. Exotic graminoid cover also showed a non-significant but slight decline in all categories, except low severity/cut and burn sites, where exotic graminoid cover increased by about 1% in 2005. These short-term data do not show support for the dieoff of wheat resulting in an ecological vacuum that is subsequently filled with exotics, but future monitoring would be useful to assess any longer-term consequences of seeding.

5. Conclusions

Our results supported the hypothesis that severely burned areas, regardless of previous forest treatments, would be characterized by higher plant canopy cover compared to low severity burn areas and have mostly ruderal colonizing species as indicator species. However, exotic plant species cover was not significantly higher on severely burned areas compared to areas classified as low burn severity. It was gratifying to observe that cover of exotic species was low in the first 3 years after the Rodeo–Chediski fire, despite our expectations to the contrary. While our study was short-term in scope during a drought period, other studies have seen an explosion of exotic species populations within 2 to 3 years following fire (Crawford et al., 2001; Floyd et al., 2006; Wienk et al., 2004). The White Mountain Apache tribal land has been managed for logging, grazing, and hunting, unlike preserves that had similar low post-fire responses by exotics. Management practices such as thinning and prescribed burning do have the potential to increase exotics due to disturbance, but tend to not have as detrimental effect as severe wildfire (Hunter et al., 2006). Our results show that seeding both post-fire and previously, contributed to the exotic species richness in the area. Although wheat largely died off within a growing season, other perennial exotic grasses which had been seeded in the area persisted and were indicator species in high severity burn sites. There were 13 exotic species found on 5% or greater of the plots in this study. Of those, three had been seeded in past management practices (*A. desertorum*, *D. glomerata*, and wheat). In addition, *B. inermis*, *E. curvula*, and *Lolium perenne* L. (perennial ryegrass) had been seeded in past years but were found on less than 5% of plots. Most of the remaining exotic species include fire-followers, or other species such as common mullein, which is listed as noxious in Colorado, but tends to be ephemeral a few years after disturbances (Sieg et al., 2003). The possibility of a sudden and rapid proliferation of existing exotic species such as Floyd et al. (2006) observed can only be addressed by continuing to monitor these sites in the future.

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References

Anderson, M.J., 2001. A new method for non-parametric multivariate analysis of variance. Aust. Ecol. 32–46.

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.

Bataineh, A.L., Oswald, B.P., Bataineh, M.M., Williams, H.M., Coble, D.W., 2006. Changes in understory vegetation of a ponderosa pine forest in northern Arizona 30 years after a wildfire. Forest Ecol. Manage. 235, 283–294.

Bond, W.J., van Wilgen, B.W., 1996. Fire and Plants. Chapman & Hall, New York, NY, pp. 34–50.

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., Andrews, J.H., 1975. Pre-Columbian purslane (*Portulaca oleracea* L.) in the new world. Nature 253, 726–727.

Clary, W.P., Ffolliott, P.F., 1996. Differences in Herbage–Timber Relationships Between Thinned and Unthinned Ponderosa Pine Stands. USDA Forest Service Research Note RM-158. Rocky Mountain Forest and Range Experiment Station, Fort Collins, CO, USA.

Cocke, A.E., Fulé, P.Z., Crouse, J.E., 2005. Comparison of burn severity assessments using differenced normalized burn ratio and ground data. Int. J. Wildland Fire 14, 189–198.

Crawford, J.A., 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.

D'Antonio, C.M., Chambers, J.C., 2006. Using ecological theory to manage or restore ecosystems affected by invasive plant species. In: Falk, D.A., Palmer, M.A., Zedler, J.B. (Eds.), Foundations of Restoration Ecology. Island Press, Washington DC, pp. 260–279.

Daubenmire, R.F., 1959. A canopy-coverage method. Northwest Sci. 33, 43–64.

DiTomaso, J.M., Kyser, G.B., Hastings, M.S., 1999. Prescribed burning for control of yellow starthistle (*Centaurea solstitialis*) and enhanced native plant diversity. Weed Sci. 47, 233–242.

Dodge, R.S., Fulé, P.Z., Sieg, C.H., in press. Dalmatian toadflax (*Linaria dalmatica*) response to wildfire in a southwestern forest. Ecoscience.

Dufrêne, M., Legendre, P., 1997. Species assemblages and indicator species: the need for a flexible asymmetrical approach. Ecol. Monogr. 67, 345–366.

Ecological Restoration Institute, 2005. Plants of Northern Arizona forests. Northern Arizona University, Flagstaff, AZ, 609 pp.

Faith, D.P., Minchin, P.R., Belbin, L., 1987. Compositional dissimilarity as a robust measure of ecological distance. Vegetatio 69, 57–68.

Finney, M.A., McHugh, C.W., Grenfell, I.C., 2005. Stand- and landscape-level effects of prescribed burning on two Arizona wildfires. Can. J. Forest Res. 35, 1714–1722.

Floyd, M.L., Hanna, D., Romme, W.H., Crews, T.E., 2006. Predicting and mitigating weed invasions to restore natural post-fire succession in Mesa Verde National Park, Colorado, USA. Int. J. Wildland Fire 15, 247–259.

Fornwalt, P.J., Kaufmann, M.R., Huckaby, L.S., Stoker, J.M., Stohlgren, T.J., 2003. Non-native plant invasions in managed and protected ponderosa pine/Douglas-fir forests of the Colorado Front Range. Forest Ecol. Manage. 177, 515–527.

Foxx, T.S., 1996. Vegetation succession after the La Mesa fire at Bandelier National Monument. In: Craig, D., Allen, (Eds.), Fire Effects in Southwestern Forests: Proceedings of the Second La Mesa Fire Symposium, 29–31 March 1994, Los Alamos, NM. USDA Forest Service, Rocky Mountain Forest and Range Experiment Station, Fort Collins, CO, pp. 47–69.

Gauch Jr., H.G., 1982. Multivariate Analysis in Community Ecology. Cambridge University Press, Cambridge.

Griffis, K.L., Crawford, J.A., Wagner, M.R., Moir, W.H., 2001. Understory response to management treatments in northern Arizona ponderosa pine forests. Forest Ecol. Manage. 146, 239–245.

Grime, J.P., 1977. Evidence for the existence of three primary strategies in plants and its relevance to ecological and evolutionary theory. Am. Nat. 111, 1169–1194.

Huffman, D.W., Moore, M.M., 2004. Responses of *Fendler ceanothus* to overstory thinning, prescribed fire and drought in an Arizona ponderosa pine forest. Forest Ecol. Manage. 198, 105–115.

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.

Hunter, M.E., Omi, P.N., Martinson, E.J., Chong, G.W., 2006. Establishment of non-native plant species after wildfires: effects of fuel treatments, abiotic and biotic factors, and post-fire grass seeding treatments. Int. J. Wildland Fire 15, 271–281.

Institute of Ecosystem Studies, 1999. Gap Light Analyzer (Computer Program). Simon Fraser University. http://www.ecostudies.org/gla.

Keeley, J.E., Lubin, D., Fotheringham, C.J., 2003. Fire and grazing impacts on plant diversity and alien plant invasions in the southern Sierra Nevada. Ecol. Appl. 13, 1355–1374.

Keeley, J.E., 2004. Ecological impacts of wheat seeding after a Sierra Nevada wildfire. Int. J. Wildland Fire 13, 73–78.

Kuenzi, A. 2006. Pre-fire treatment effects and understory plant community response on the Rodeo–Chediski fire, Arizona. M.S. Thesis, School of Forestry, Northern Arizona University, Flagstaff, Arizona.

Lake, J.C., Leishman, M.R., 2004. Invasion success of exotic plants in natural ecosystems: the role of disturbance, plant attributes and freedom from herbivores. Biol. Conserv. 117, 215–226.

Laughlin, D.C., Bakker, J.D., Stoddard, M.T., Daniels, M.L., Springer, J.D., Gildar, C.N., Green, A.M., Covington, W.W., 2004. Toward reference conditions: wildfire effects on flora in an old-growth ponderosa pine forest. Forest Ecol. Manage. 199, 137–152.

McCune, B., Mefford, M.J., 1999. PC-ORD (Computer Program). Multivariate Analysis of Ecological Data, Version 4. MJM Software Design, Gleneden Beach, Oregon.

Milchunas, 2006. Responses of plant communities to grazing in the southwestern United States. USDA For. Serv. Gen. Tech. Rep. RMRS-GTR-169., Rocky Mountain Research Station, Fort Collins, CO, 126 pp.

Mitchell, W.R., 1981. Soil Survey of Fort Apache Indian Reservation, Arizona, Parts of Apache, Gila, and Navajo Counties. USDA Soil Conservation Service and United States Department of Interior Bureau of Indian Affairs in Cooperation with Arizona Agricultural Experiment Station.

Moore, M.M., Casey, C.A., Bakker, J.D., Springer, J.D., Fulé, P.Z., Covington, W.W., Laughlin, D.C., 2006. Herbaceous vegetation responses (1992–2004) to restoration treatments in a ponderosa pine forest. Rangeland Ecol. Manage. 59, 135–144.

Sabo, K.E., 2006. Overstory and understory production in varying stand structural types in northern Arizona ponderosa pine forests. M.S. Thesis, School of Forestry, Flagstaff, AZ, Northern Arizona University.

Sieg, C.H., Phillips, B.G., Moser, L.P., 2003. Exotic Invasive Plants. In: Friederici, P. (Ed.), Ecological Restoration of Southwestern Ponderosa Pine Forests. Island Press, Washington, pp. 251–267.

Soil Survey Staff, 2006. Natural Resources Conservation Service, United States Department of Agriculture. Official Soil Series Descriptions [online]. Available from http://soils.usda.gov/technical/classification/osd/index.html [accessed 17 July 2006].

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, Flagstaff, AZ, Northern Arizona University. Available at http://library.for.nau.edu:8443/handle/2175/67.

Swope, S.M., 2003. Exotic plant invasions in relation to diversity, limiting resources and livestock grazing in Arizona grasslands. M.S. Thesis, School of Forestry, Flagstaff, AZ, Northern Arizona University.

USDA Forest Service, 2002. Rodeo–Chediski Fire Effects Summary Report: Apache-Sitgreaves National Forests. USDA Forest Service, Washington, DC.

USDA, NRCS 2006. The PLANTS Database (http://plants.usda.gov, 29 November 2006). National Plant Data Center, Baton Rouge, LA 70874-4490, USA.

Welsh, S.L., Atwood, N.D., Goodrich, S., Higgins, L.C., 1993. A Utah Flora, second ed. Brigham Young University, Provo, UT.

Westerling, A.L., Hidalgo, H.G., Cayan, D.R., Swetnam, T.W., 2006. Warming and earlier spring increase western U.S. forest wildfire activity. Science 313, 940–943.

Wienk, C.L., Sieg, C.H., McPherson, G.R., 2004. Evaluating the role of cutting treatments, fire and soil seed banks in an experimental framework in ponderosa pine forests of the Black Hills, South Dakota. Forest Ecol. Manage. 192, 375–393.

Wolfson, B.A.S., Kolb, T.E., Sieg, C.H., Clancy, K.M., 2005. Effects of post-fire conditions on germination and seedling success of diffuse knapweed in northern Arizona. Forest Ecol. Manage. 216, 342–358.

Young, R.P., 1983. Fire as a vegetation management tool in rangelands of the Intermountain Region. In: Monsen, S.B., Shaw, N. (compilers). Managing Intermountain rangelands—Improvement of Range and Wildlife Habitats: Proceedings; 1981 September 15–17; Twin Falls, ID; 1982 June 22–24; Elko, NV. Gen. Tech. Rep. INT-157. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station, pp. 18–31.