

PRE-FIRE TREATMENT EFFECTS AND UNDERSTORY PLANT COMMUNITY
RESPONSE ON THE RODEO-CHEDISKI FIRE, ARIZONA

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ABSTRACT

PRE-FIRE TREATMENT EFFECTS AND UNDERSTORY PLANT COMMUNITY RESPONSE ON THE RODEO-CHEDISKI FIRE, ARIZONA

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High severity wildfires have been increasing across southwestern ponderosa pine forests in recent decades. As the effects of wildfire become more widespread across the landscape, the need for information about the ecological effects of fire on understory vegetation is mounting. We investigated understory plant community response to the Rodeo-Chediski fire by conducting parallel studies on the White Mountain Apache Tribal lands (WMAT) and the Apache-Sitgreaves National Forests (A-S). We estimated plant canopy cover by species and quantified total plant species richness on 1000m² plots. We established 71 plots on WMAT lands. Plots were stratified by fire severity (low and high) and pre-fire treatment (cut/burned and untreated). We found significantly higher plant cover on areas that were burned by severe fire, but did not detect significant differences due to pre-fire treatment. There was no significant difference in cover of exotic species between high and low severity sites. Indicator species were primarily early successional species or species that were included in the post-fire seed mix.

We established 84 plots on the A-S. This study consisted of 7 paired stands of treated (pre-fire fuel reduction) and untreated sites. These sites had been established by the USDA Forest Service in 2002 after the fire to assess effectiveness of fuel-reduction treatments in altering fire behavior. We found significantly higher plant cover on areas that had not been treated, and had therefore burned with higher severity. There was no

significant difference in cover of exotic species between treated and untreated stands.

Indicator species were primarily early successional species or species that were included in the post-fire seed mix.

In both studies we found a limited response of exotics, which is surprising given the history of active management by both land agencies. Other studies have shown higher exotic species presence following severe wildfires in areas with histories of active management that included practices such as logging, grazing, and seeding. The pre-fire plant community must not have had large populations of exotic species, despite past management practices to result in a native-dominated post-fire community. Whether the few exotic species we observed, including a number of species seeded following the fire, persist and spread in the post-fire plant community remains to be seen.

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PREFACE

This thesis contains two manuscript chapters intended for publication. The manuscripts are Chapter 2: “Limited response of exotic species after a severe landscape-scale wildfire” and Chapter 3: “Influence of pre-fire fuel reduction treatments on understory response following a severe wildfire, Apache-Sitgreaves National Forests, USA.” These chapters use “we” instead of “I” because they include co-authors. A list of the literature cited appears at the end of each chapter. The text has been edited to avoid redundancy wherever possible.

Chapter 1

Introduction

Across the southwestern United States, fires are increasing in severity and frequency (Westerling *et al.* 2006). As the effects of wildfire become more widespread across the landscape, the need for information about the understory plant community response is mounting (Abella 2004). Years of fire suppression have resulted in ponderosa pine (*Pinus ponderosa* P.& C. Lawson var. *scopulorum* Engelm.) forests that have much higher tree density than in pre-European settlement conditions (Covington & Moore 1994, Covington *et al.* 1997). Understory plants make up the majority of the plant biodiversity of southwestern ponderosa pine forests (Moore *et al.* 2006). However, in dense forests little sunlight reaches the understory, and large amounts of leaf litter prevent plant growth. As wildfire returns to the landscape, openings in the canopy and bare mineral soil allow new opportunity for plants to colonize.

Disturbance ecology

Grime (1977) outlined three strategies for plant establishment in response to stress and disturbance. He defines stress as “the external constraints which limit the rate of dry-matter production of all or part of the vegetation.” Stress is the result of lack of water, nutrients, sunlight, any other limiting factor, or the addition of some factor, such as a pollutant that inhibits growth. Grime defines disturbance as “destruction of vegetation,” and can include human causes such as plowing or mowing, and natural events such as floods, windstorms, and of course, fire. He outlined the ruderal (R-selection), competitive (C-selection), and stress-tolerant (S-selection) strategies of plant evolution. Ruderal species, or R-selected species, are adapted to environments with high disturbance but low

stress, and therefore colonize areas that are potentially high in nutrients and other resources after disturbance. Ruderal species typically have a short life-span and invest many resources into reproduction, for example, by high seed production. Competitive species are adapted to low disturbance and low stress. This strategy maximizes vegetative growth. Stress-tolerant species are adapted to high stress and low disturbance, and therefore do not invest many resources into vegetative growth or high reproductive rates. S-selected species are adapted to tolerate unfavorable conditions and generally have longer life-spans. Grime also related these ideas about plant adaptation strategies to the classic ecological concept of r- and K- selected species (Pianka 1970). The ruderal species strategy corresponds to r-selected species, sharing the characteristics mentioned above, and the stress-tolerant strategy corresponds to K-selection. The competitive strategy falls between the two extremes. These strategies describe a gradient of plant traits, and invasive exotic plants may fall anywhere along that gradient. However, the characteristics of the ruderal or r-selected species are more likely found in invasive plants (D'Antonio & Chambers 2006). Lake & Leishman (2004) stated that some form of disturbance is necessary for invasion of exotics into natural ecosystems, and that the type of disturbance determines how successful the invasion will be. Many exotic species use this strategy to invade bare mineral soil to fill the gaps left by the consumed vegetation, as do some early successional native species. Commonly, these are annual or biennial species.

Effects of exotic species

World-wide, exotic species invasion threatens biodiversity. It is the second greatest threat to threatened and endangered species in the U.S., the first being habitat

loss (Wilcove *et al.* 2000). Exotic, invasive plants can affect fire regimes by altering fuel loads, fuel continuity, and fuel packing ratios (Brooks *et al.* 2004). Because exotic plants commonly invade after disturbances such as fire, some species can become part of a feedback loop, creating more fuels that can alter fire regimes. High severity fire has been shown to increase some exotic species. Crawford *et al.* (2001) found twice as much cover of exotic species on high severity burns as on moderate burns on three wildfires in northern Arizona.

Management

Many studies have shown that fuel reduction treatments make significant impacts on fire behavior and severity (Finney *et al.* 2005, Pollet & Omi 2003, Strom & Fulé in press). However, the disturbance created by fuel reduction treatments also has an effect on the understory (Sieg *et al.* 2003). Fuel reduction treatments can increase exotic plant cover (Dodson & Fiedler 2006, Griffis *et al.* 2001, Keeley 2006, Wienk *et al.* 2004), possibly setting the stage for increased exotic cover after wildfire through increased exotics in the seed bank and propagule pressure from surrounding plant communities. However, the reduction in fire severity is likely a more beneficial trade-off as it has been shown that wildfires have greater potential for exotic species invasion than do fuel treatments (Griffis *et al.* 2001, Hunter *et al.* 2006). Griffis *et al.* (2001) found >40% more exotic forbs on sites that had been severely burned by wildfire than on those that were thinned and prescribed burned. Other factors can increase exotics in an ecosystem as well. Grazing, past seeding practices, and road building are all avenues for exotic species introduction. Huisinga *et al.* (2005) reported a native species-dominated understory after an intense prescribed fire on the North Rim of Grand Canyon National Park, which they

attributed to limited grazing and no logging. The same has been found in other lands managed for preservation. Foxx (1996) found almost no exotic species in a 16-year study of the 1977 La Mesa fire at Bandelier National Monument. However, there have been contrasting findings regarding the role of management in promoting exotic invasion. Fornwalt *et al.* (2003) compared two forested areas in the Colorado Front Range. One had been actively managed by the Forest Service, including logging, grazing, and prescribed burning. The other area had been almost completely undisturbed since the beginning of the 20th century. They found no differences in exotic species richness or cover on the two areas.

Seeding

Beyers (2004) stated that effective erosion control is rarely accomplished by post-fire seeding. She noted that seeded grass species can displace native annual forbs, and reduce tree seedling survival. Barclay *et al.* (2004) examined the effects of seeding *Lolium multiflorum* Lam. (ryegrass), which is widely used by land management agencies as part of Burned Area Emergency Rehabilitation. They found cover of native forbs was higher in non-seeded areas and survivorship of conifer seedlings decreased in seeded areas. They suggested that the short-term presence of ryegrass may affect the long-term succession of post-fire plant communities and that seeding may introduce other exotic species. Keeley (2006) cited one example of the introduction of over one billion *Bromus tectorum* L. (cheatgrass) seeds onto the burned areas of the 2000 Cerro Grande fire in New Mexico by aerial seeding. *Triticum aestivum* L. (common wheat) is another cereal grass that has been used for erosion mitigation due to its non-persistent nature. However, the effects are the same in that its application can result in decreased native forb cover

and pine seedling recruitment (Keeley 2004). In addition, seeding with nonpersistent grasses may create an “ecological vacuum” in the second year post-fire that can lead to invasion by colonizing exotics (Keeley 2006).

Research goals

The Rodeo-Chediski fire burned 189,650 ha in east-central Arizona from June 18 to July 7, 2002, leaving behind a mosaic of burn severities on 113,700 ha of the White Mountain Apache tribal land and 67,670 ha of the Apache-Sitgreaves National Forests. This fire exhibited some of the most extreme fire behavior ever seen in the Southwest (USDA Forest Service, 2002). However, even under such extreme conditions, prior fuel reduction treatments resulted in areas of low severity burns (Cram & Baker 2003, Finney *et al.* 2005, Strom & Fulé in press). High severity areas were seeded after the fire, using a combination of native and exotic species. The landscape of treated and untreated areas, combined with high and low burned severities allows for a unique opportunity to investigate a “natural experiment” on the short-term understory species response.

There is limited information available on fuel reduction treatments and post-wildfire effects on understory communities. The goal of this project was to quantify plant community composition under different severities and treatments. We broke our research into two portions, investigating the White Mountain Apache Tribal lands and the Apache-Sitgreaves National Forests separately. For each area of the study we were interested in total plant canopy cover and richness, cover and richness of exotic species, and the response of various plant groups. We were also interested in the response of the seeded species. Data were collected from July to early September (to coincide with the monsoon season) in 2004 and 2005. We analyzed data using both univariate and multivariate

methods as recommended by Abella (2004). We ran Indicator Species Analyses to provide further insight as to which species were dominant under the differing conditions. Chapter 2 focuses on the work done on the White Mountain Apache tribal land. Chapter 3 describes the study on the Apache-Sitgreaves National Forests.

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Chapter 2

Limited response of exotic species after a severe landscape-scale wildfire

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 the White Mountain Apache tribal land (WMAT). In 2004 and 2005, we measured cover and richness in the understory plant community in areas of high and low severity in each of two treatments: (1) tree cutting and prescribed burning, or (2) left untreated, in the 11 years prior to the wildfire. Cutting and burning reduced the degree of severe burning, but severely burned areas in either treatment were similar. Total understory plant canopy cover was significantly higher in areas of high severity ($p = .0002$ in 2004 and $p = .0001$ in 2005). Overall, response of exotic species was surprisingly low at less than 3% cover 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 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 all indicators of high severity were seeded or early successional species. Indicators of low severity included several perennial bunchgrasses. While our results show that wheat declined quickly and exotic plants were uncommon, whether the die-off of wheat created a vacuum for exotic plant invasion in subsequent years can only be addressed by continuing to monitor these sites.

Introduction

Increasing numbers of large wildfires in ponderosa pine 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 (Brooks *et al.* 2004, Kerns *et al.* 2006, Sieg *et al.* 2003). 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 & Leishman 2004).

In southwestern ponderosa pine forests, Griffis *et al.* (2001) examined 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 early successional native species such as *Conyza canadensis* (L.) Cronq. (Canadian horseweed), and cautioned about using prescribed burns as a management tool if exotic species removal is the desired outcome. In contrast, a study by Huisinga *et al.* (2005) found that total plant species richness and abundance both increased after a high intensity wildfire in Grand Canyon National Park, mainly due to an increase in native plants. 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 1930's), so there were few or no exotic species seeds present in the seed bank. However, 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 rarity of exotic species in their study. A 16-year study by Foxx (1996) on the 1977 La Mesa fire in Bandelier National Monument, NM also found few exotic species across the entire burned area, perhaps because of the "preservation" management strategy of the National Park Service. One seeded species, slender wheatgrass (*Elymus trachycaulus* (Link) Gould ex Shinnars), was initially a dominant species after the fire but dropped out within a decade. Early successional species, such as fetid goosefoot (*Chenopodium graveolens* Willd.), were also dominant right after the fire and eventually dropped out.

The largest severe wildfire in the Southwest, the Rodeo-Chediski fire, 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 (henceforth WMAT). The fire burned through vegetation types including chaparral, piñon-juniper woodland, 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 yrs pre-fire) treatments (Finney *et al.* 2005, Strom 2005).

We investigated the effect of the pre-fire management practices and fire severity on the plant community after a large, intense wildfire. Given the long history of timber management on the WMAT 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 richness of early successional species; and 3) higher exotic species richness and plant canopy cover, compared to areas classified as low burn severity. We also expected that species seeded post-burn would attain higher cover in high severity areas, and that *Triticum aestivum* L. (henceforth referred to as common wheat), seeded post-fire, would be a prominent species in seeded areas, but would not persist over time.

Methods

Site selection

The WMAT 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 and average annual total snowfall is 97.3 cm. Precipitation during the years of data collection was lower than the 55-year averages. The area received 33.3 cm of precipitation in 2004 and 40.7 cm in 2005. Total snowfall in winter of 2003/2004

was 50.3 cm and 22.9 cm in winter of 2004/2005. Elevation for the study area ranges from 2,000-2,295 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 and two levels of pre-fire forest management practices (henceforth referred to as treatments). Fire severity was determined from a remotely sensed Differenced Normalized Burn Ratio (Δ NBR) map. The thresholds for low and high severity were assigned by Strom (2005) based on field measurements of another fire in northern Arizona by Cocke *et al.* (2005). If we arrived at a site classified as ‘low severity’ and did not see any living trees, we rejected it. If a site classified as ‘high severity’ had many living trees, then it was rejected. The treatment categories were ‘cut and burned’ within 10 years prior to the fire or ‘no treatment’ within the same period. 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). The seed mix used on WMAT 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*, western wheatgrass (*Pascopyrum smithii* (Rydb.) A. Löve), switchgrass (*Panicum*

virgatum L.), green needlegrass (*Nassella viridula* (Trin.) Barkworth), mountain brome (*Bromus maritimus* = *Bromus carinatus* H. & A.) (Welsh *et al.* 1993), sideoats grama (*Bouteloua curtipendula* (Michx.) Torr.), sand dropseed (*Sporobolus cryptandrus* (Torr.) Gray), plains coreopsis (*Coreopsis tinctoria* Nutt.), purple prairie clover (*Dalea purpurea* Vent.), blue flax (*Linum lewisii* Pursh), and black-eyed susan (*Rudbeckia hirta* L.) (J.A. Youtz, pers. comm., 2003).

Measurement

Six study sites, with three plots each, were measured in each severity and treatment combination. Plot centers were a subset of those used in 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 x 31.7 m square, approximately 1000 m². We measured plant canopy cover in 20- by 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 (Table 1) (Daubenmire 1959). We also estimated cover of litter, rocks, logs, bare ground, and moss. Plant nomenclature and nativity classification follows USDA-NRCS PLANTS Database (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 & McAndrews (1975). We lumped species to the generic level when vegetative characteristics were insufficient to identify at species level. In addition, we quantified plant species richness for the entire plot by recording all species that occurred within each 1000-m² 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.

Statistical Methods

We analyzed the data using DISTLM (Anderson 2001 & 2004), a permutation procedure software which analyzes multivariate data, so it was an appropriate choice to test differences in the plant community. For consistency, we also used DISTLM 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 & Mefford 1999). 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 X percent canopy cover. Species with $p < 0.05$ and an Indicator Value > 25 (Dufrêne & Legendre 1997) were identified as indicator species.

Results

In high severity areas across both treatments tree density declined by 95% and basal area declined by 90% after the fire (Table 2). In low severity areas tree density declined by 40% and basal area declined by 18%. The pre-fire tree density was mainly composed of ponderosa pine (*Pinus ponderosa* P.& C. Lawson var. *scopulorum* Engelm.), but also had Douglas-fir (*Pseudotsuga menziesii* (Mirbel) Franco), Gambel oak (*Quercus gambelii* Nutt.), white fir (*Abies concolor* (Gord. & Glend.) Lindl. ex Hildebr.) and junipers (*Juniperus osteosperma* (Torr.) Little and *Juniperus deppeana* Steud.). Post-fire high severity areas had no living *A. concolor* or *P. menziesii* trees. The tree density of *P. ponderosa* decreased by more than 75%, *Juniperus* spp. decreased by 30%, and *Q. gambelii* decreased by 40%. In low severity areas, the tree density of *A. concolor* decreased by 70%, *Juniperus* spp. decreased by 6%, *P. ponderosa* decreased by 30%, *P. menziesii* decreased by 40%, and *Q. gambelii* decreased by 75%. Canopy openness post-burn was >65% in high severity areas and <46% in low severity areas (Table 2).

We found a total of 310 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. Total plant species richness was not significantly different for any combination of severity and treatment in either year (Table 3, 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 3).

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 3, 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,b). There were 35 exotic species overall, but 13 were found on at least 5% of the plots and were included in analyses (Table 4). 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. 2 c-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 in both years for severity effect ($p = 0.0001$ in both years). The cover by species was significantly different for treatment effect in 2004 ($p = 0.044$), but not significant in 2005 ($p = 0.10$). This appeared to be driven by one species, *Robinia neomexicana* Gray (New Mexico locust), which had nearly 32 times more cover in untreated areas than in areas that were thinned and burned. To test differences in the composition of the plant community we analyzed the presence/absence of species and found significant differences for both years for both treatment (2004, $p = 0.0036$; 2005, $p = 0.0046$) and severity effects ($p = 0.0001$ both years). In each case, the difference was driven by very few species (Tables 5a,b).

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 (Figure 3a,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 in the permutation procedure.

We identified 9 indicator species for high severity in 2004 and 10 in 2005 (Table 6). *B. carinatus*, *E. trachycaulus*, and common wheat were seeded species. *Chenopodium album* L. (lambsquarters), *C. graveolens*, *Cirsium wheeleri* (Gray) Petrak (Wheeler's thistle), and *C. 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 & 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 (Figure 4). In 2004, wheat was highest in high severity areas that had received no pre-fire treatment. In low 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.

Discussion

Across the entire Rodeo-Chediski fire on tribal lands, treatment substantially reduced fire behavior over the landscape (Finney *et al.* 2005). Over 60% of the area that had been thinned and prescribed burned within a decade of the fire burned with low severity, as opposed to only 20% of the area that had not been treated for the same time period (Strom 2005). However, comparing areas of high and low severity *within* treated and untreated areas, the hypothesis that fire severity would play a more important role in the post-wildfire plant community than treatment was upheld to some degree. 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 $8 \text{ m}^2 \text{ ha}^{-1}$ or less to result in an increase in understory production. Clary & Ffolliott (1966) found an understory response at $5\text{-}18 \text{ m}^2 \text{ ha}^{-1}$. Basal area in the high severity sites after the fire was $1.9 \text{ m}^2 \text{ ha}^{-1}$ in untreated sites and $0.9 \text{ m}^2 \text{ ha}^{-1}$ in treated sites, clearly low enough to result in an understory response. In low severity areas basal area was $21.6 \text{ m}^2 \text{ ha}^{-1}$ and $14.0 \text{ m}^2 \text{ ha}^{-1}$, respectively. Differences in post-fire overstory structure, combined with other factors such as variable precipitation (Moore *et al.* 2006), likely contributed to differences in understory cover between high and low severity sites.

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. Some studies in more actively managed landscapes, including national forests, have shown higher exotic cover. 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 7). Husinga *et al.* (2005) found 28% total plant cover six 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 cover of native 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 thinning and grazing, but the exotic species cover is similar with <1% cover in both 2004 and 2005.

Our results contrast with Crawford *et al.* (2001), who found twice as much exotic cover on high severity burns as on moderate burns. However, a substantial portion of the exotic cover in the Crawford *et al.* (2001) study was *C. album*. Husinga *et al.* (2005) recognized the difficulty in distinguishing *C. album* from other native, broad-leaved *Chenopodium* species (e.g. *C. fremontii*), and considered the possibility of misidentification of *C. album* when discussing the results of Crawford *et al.* (2001). In addition, Crawford *et al.* (2001) classified *Conyza canadensis* and *Chenopodium graveolens* as exotics. We classify both species as native based on USDA-NRCS PLANTS Database (2006). Even with these changes, mean exotic plant cover was twice as high on high severity burns compared to moderate burns (59% vs 20%, respectively) in the Crawford *et al.* (2001) paper. In our study, all broad-leaved *Chenopodium* were called *C. album*, but were easily distinguished from *C. graveolens*, *C. botrys*, and *C. leptophyllum*, which were the other species found on our plots. If another native, broad-

leaved *Chenopodium* were misidentified as *C. album*, then it would have resulted in an inflated measurement in exotic forbs.

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, but *Linaria dalmatica* (L.) P. Mill. Ssp. *dalmatica* (Dalmatian toadflax) and *Salsola tragus* L. (Russian thistle) were not found at all. Most exotic species found are not considered noxious in southwestern states, were found in very low cover, or were seeded (Table 4, Fig 2 a,b).

Our finding that cover of exotic plants did not differ significantly between treatments is in agreement with Hunter *et al.*’s (2006) results. While they recognized there is some potential for fuel-treatments to increase propagule pressure of exotic plants, their results did not support this. However, in contrast to our study, they found fire severity to be a major predictor of exotic species establishment.

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 1990’s, when the policy was revised to using only native species. In the mid 1990’s 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. (orchardgrass), 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 greatly affects understory plant communities. 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 precipitation were less than the 55-year averages. Perhaps insufficient precipitation, especially snowfall, played a role in limiting the response of exotic, annual species.

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 6) five were specifically seeded in high severity areas (*B. curtipendula*, *B. 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 & Moore, 2004). Both *C. album* and *C. 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*, *C. wheeleri*, *C. canadensis*, and *P. macounii* are disturbance tolerant or dependent on disturbances for their 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 *et al.* 1996) or from below the soil surface where they are protected (Young 1983). In addition, *P. neomexicana* was an indicator of low severity in one 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 (Table 5 a, b). 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 creates "an ecological vacuum," which can 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 (SE = 0.87) to 2.1% cover (SE = 1.47) in 2005. All other combinations of severity and treatment showed slight declines in exotic forb cover in 2005. Exotic graminoid cover also declined slightly in all categories, except low severity/cut & burn sites, where exotic graminoid cover was absent in 2004 and averaged 1.2% 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 longer -term data may lend support to that hypothesis.

Conclusions

While it may be gratifying to observe that the response of exotic species was unexpectedly low in the first two years following the Rodeo-Chediski fire, we see few concrete reasons why the exotic response was so low. The WMAT 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). Therefore fuel reduction treatments are important for reducing the overall severity of wildfire (Finney *et al.* 2005, Savage & Mast 2005, Strom 2005). Finally, we recommend that managers strongly consider not using exotic species in post-wildfire seeding projects. 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, and other species such as common mullein, which is listed as noxious in Colorado, but tends to be ephemeral after disturbances (Sieg *et al.* 2003). While our results show that common wheat declined quickly, the cautions about wheat seeding raised by Keeley (2004) about the die-off of wheat creating a vacuum for exotic plant invasion in subsequent years can only be addressed by continuing to monitor these sites in the future.

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Table 1. Plant canopy cover classes used to classify percent cover within a 20- by 50-cm quadrat (modified from Daubenmire 1959).

Cover Class	Range of Coverage	Midpoint
T	0 – .999 %	.5
1	1 – 5 %	3
2	6 – 25 %	15.5
3	26 – 50 %	38
4	51 – 75 %	63
5	76 – 95 %	85.5
6	96 – 100 %	98

Table 2. Overstory characteristics of sites on White Mountain Apache Tribal lands by pre-fire treatments and severities. 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).

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	No Trt	66.9 (2.3)	44.2 (35.9)	508.5 (137.5)	1.9 (1.5)	15.1 (3.1)
	Cut & Burn	65.1 (1.6)	9.9 (4.6)	632.5 (168.7)	0.9 (0.4)	12.0 (1.7)
Low	No Trt	39.4 (2.4)	647.5 (147.6)	937.6 (184.1)	21.6 (2.8)	25.5 (3.0)
	Cut & Burn	45.6 (2.1)	343.0 (65.3)	728.7 (222.6)	14.0 (1.8)	17.8 (2.8)

Table 3. A summary of the permuted p-values for DISTLM tests for univariate plant canopy cover and richness, by category. An asterisk indicates significance.

Cover						
Category	Severity		Treatment		Interaction	
	2004	2005	2004	2004	2004	2005
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						
Category	Severity		Treatment		Interaction	
	2004	2005	2004	2005	2004	2005
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

Table 4. Summary of observed exotic species by fire severity. Classification of ‘noxious’ is based on designation in at least one of four southwestern states, according to Sieg *et al.* (2003). Species in the ‘seeded’ category may have been seeded throughout the known history of the WMAT 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.

Severity	Species Present ¹		
High	Seeded	Noxious	Other
	<i>Agropyron desertorum</i> <i>Bromus inermis</i> ² <i>Dactylis glomerata</i> <i>Eragrostis curvula</i> <i>Lolium perenne</i> <i>Triticum aestivum</i>	<i>Bromus tectorum</i> <i>Cirsium vulgare</i> <i>Convolvulus arvensis</i> <i>Erodium cicutarium</i> <i>Onopordum acanthium</i> <i>Verbascum thapsus</i>	<i>Bromus japonicus</i> <i>Chenopodium album</i> <i>Echinochloa crus-galli</i> <i>Kochia scoparia</i> <i>Lactuca serriola</i> <i>Lolium arundinaceum</i> <i>Malva neglecta</i> <i>Medicago lupulina</i> <i>Melilotus officinalis</i> ² <i>Poa compressa</i> ² <i>Poa pratensis</i> <i>Polygonum aviculare</i> <i>Polygonum convolvulus</i> ² <i>Polypogon viridis</i> <i>Rumex acetosella</i> <i>Setaria viridis</i> <i>Sonchus asper</i> <i>Taraxacum officinale</i> <i>Tragopogon dubius</i>
Low	Seeded	Noxious	Other
	<i>Agropyron desertorum</i> <i>Bromus inermis</i> <i>Dactylis glomerata</i> <i>Eragrostis curvula</i> <i>Triticum aestivum</i>	<i>Bromus tectorum</i> <i>Cirsium vulgare</i> <i>Convolvulus arvensis</i> <i>Verbascum thapsus</i>	<i>Bromus japonicus</i> <i>Chenopodium album</i> <i>Eragrostis lehmanniana</i> <i>Lactuca serriola</i> <i>Lolium arundinaceum</i> ² <i>Medicago lupulina</i> <i>Melilotus officinalis</i> ² <i>Poa compressa</i> ² <i>Poa pratensis</i> <i>Polygonum aviculare</i> <i>Polygonum convolvulus</i> ² <i>Rumex acetosella</i> <i>Rumex crispus</i> <i>Taraxacum officinale</i> <i>Tragopogon dubius</i>

¹Species 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.

²Species found only in 2005.

Table 5a. Species present only in untreated or only in cut and burned sites.

Species found only in Untreated Sites	2004	2005
<i>Achnatherum nelsonii</i> (Scribn.) Barkworth	X	
<i>Amaranthus hybridus</i> L.		X
<i>Amaranthus powellii</i> S. Wats.		X
<i>Ambrosia acanthicarpa</i> Hook.	X	X
<i>Brickellia grandiflora</i> (Hook.) Nutt.	X	
<i>Cercocarpus montanus</i> Raf.	X	X
<i>Frasera speciosa</i> Dougl. ex Griseb	X	X
<i>Lathyrus laetivirens</i> Greene ex Rydb.	X	
<i>Mahonia repens</i> (Lindl.) G. Don	X	X
<i>Opuntia</i> sp.	X	
<i>Portulaca oleracea</i> L.		X
<i>Rhus glabra</i> L.	X	X
<i>Rhus trilobata</i> Nutt.	X	X
<i>Townsendia exscapa</i> (Richards.) Porter		X
Species found only in Cut & Burned Sites	2004	2005
<i>Amaranthus powellii</i> S. Wats.	X	
<i>Antennaria parvifolia</i> Nutt.	X	
<i>Chamaesyce fendleri</i> (Torr. & Gray) Small		X
<i>Dactylis glomerata</i> L.	X	
<i>Dyssodia papposa</i> (Vent.) A.S. Hitchc.		X
<i>Ipomoea costellata</i> Torr.		X
<i>Muhlenbergia minutissima</i> (Steud.) Swallen	X	
<i>Piptochaetium pringlei</i> (Beal) Parodi	X	

Table 5b. Species present only in high severity sites or only in low severity sites.

Species found only in High Severity Sites	2004	2005
<i>Agropyron desertorum</i> (Fisch. ex Link) J.A. Schultes		X
<i>Amaranthus hybridus</i> L.		X
<i>Ambrosia acanthicarpa</i> Hook.		X
<i>Dyssodia papposa</i> (Vent.) A.S. Hitchc.	X	X
<i>Mahonia repens</i> (Lindl.) G. Don	X	X
<i>Pennellia longifolia</i> (Benth.) Rollins		X
<i>Portulaca oleracea</i> L.		X
<i>Sporobolus compositus</i> (Poir.) Merr.	X	
Species found only in Low Severity Site	2004	2005
<i>Desmodium grahamii</i> Gray	X	
<i>Microsteris gracilis</i> (Hook.) Greene	X	
<i>Pseudotsuga menziesii</i> (Mirbel) Franco	X	
<i>Quercus turbinella</i> Greene		X

Table 6. Indicator species by year and level of fire severity.

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

*"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.

Table 7: A comparison of cover and richness of exotic plant species following severe wildfires in the Southwest.

Site Author (year)	Time since fire	Notes	# of plots	Method for quantifying cover	Cover of exotics* (area sampled)	Richness of exotics* (area sampled)
Wht. Mtn. Apache Tribal land, AZ Kuenzi (2006)	2 yrs 3 yrs	This study.	71	cover class	0.9% in 2004 0.7% in 2005 (2m ² =20*.1m ²)	35, 13 on >5% plots [†] (≈ 1000m ²)
Apache-Sitgreaves NF, AZ Kuenzi (2006)	2 yrs 3 yrs	Also Rodeo-Chediski fire.	84	cover class	0.3% in 2004 1.6% in 2005 (2m ² =20*.1m ²)	38, 14 on >5% plots [†] (≈ 1000m ²)
Kaibab and Coconino NF, AZ Crawford <i>et al.</i> (2001)	2 yrs	Data were pooled from 3 wildfires	34	cover class	59% in High sev* 20% in Mod sev* (2.4m ² =24 * 0.1m ²)	25* (2.4m ² =24 * 0.1m ²)
Coconino NF, AZ Griffis <i>et al.</i> (2001)	<5 yrs	Data based only on the 10 wildfire plots	10	11 min search and abundance rating	≈ 50% forbs, ≈ 8% graminoids (375 m ²)	10 (375 m ²)
Grand Canyon NP, AZ Husinga <i>et al.</i> (2005)	6 yrs 8 yrs	This was an intense prescribed fire	60	point intercept	0.7% in 1999 1% in 2001 (2, 50-m transects, 166 pts ea.)	4 (1000m ²)
Bandelier Nat'l Monument, NM Barclay <i>et al.</i> (2004)	2 yrs	Data based only on non- seeded plots in 1998	49	line intercept	≈ 18% in High sev* ≈ 15% in Mod sev* (50-m transect)	?
Bandelier Nat'l Monument, NM Foxx (1996)	8 yrs	Data is based only on the first 8 years of data collection after the fire	?	cover class	≈ 0.1% (50 quadrats)	3 (1858.1 m ²)

*Tallied for the entire study, not per plot or per m².

*These numbers have been adjusted from the original study based on Husinga *et al.* (2005). *C. album* was subtracted from the total. *C. canadensis* was also subtracted due to misclassification as an exotic.

†All analysis was done on species occurring on over 5% of the plots.

• *C. canadensis* was classified as an exotic and included in this measure.

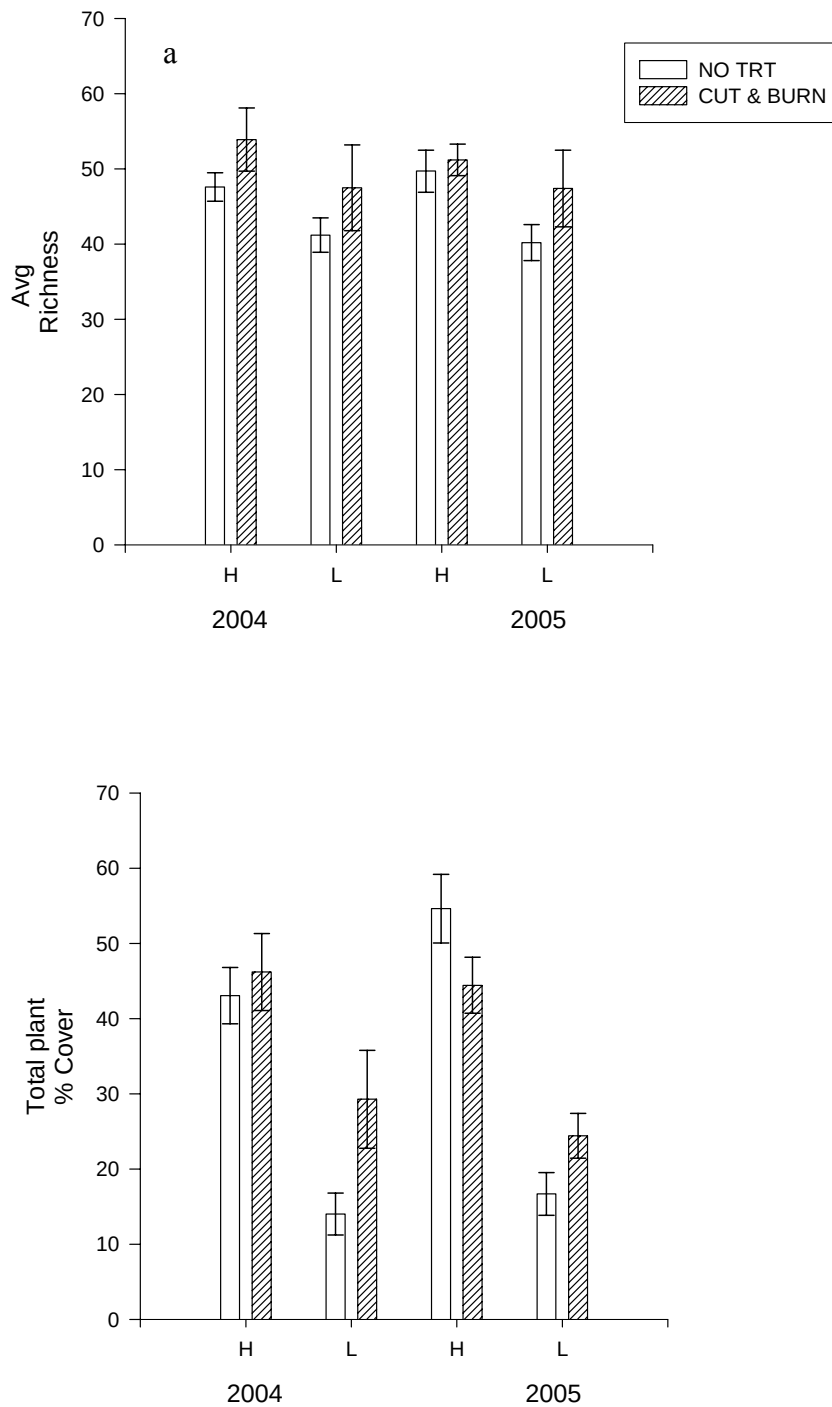


Figure 1. a) Average (+SE) richness on 1000m² plot by fire severity (H= high, L= low), treatment, and year on the WMAT plots. No significant difference was found for treatment or severity in either year. b) Average (+SE) plant canopy cover on 20- by 50- cm quadrats by fire severity (H= high, L= low), treatment, and year on the WMAT plots. 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.

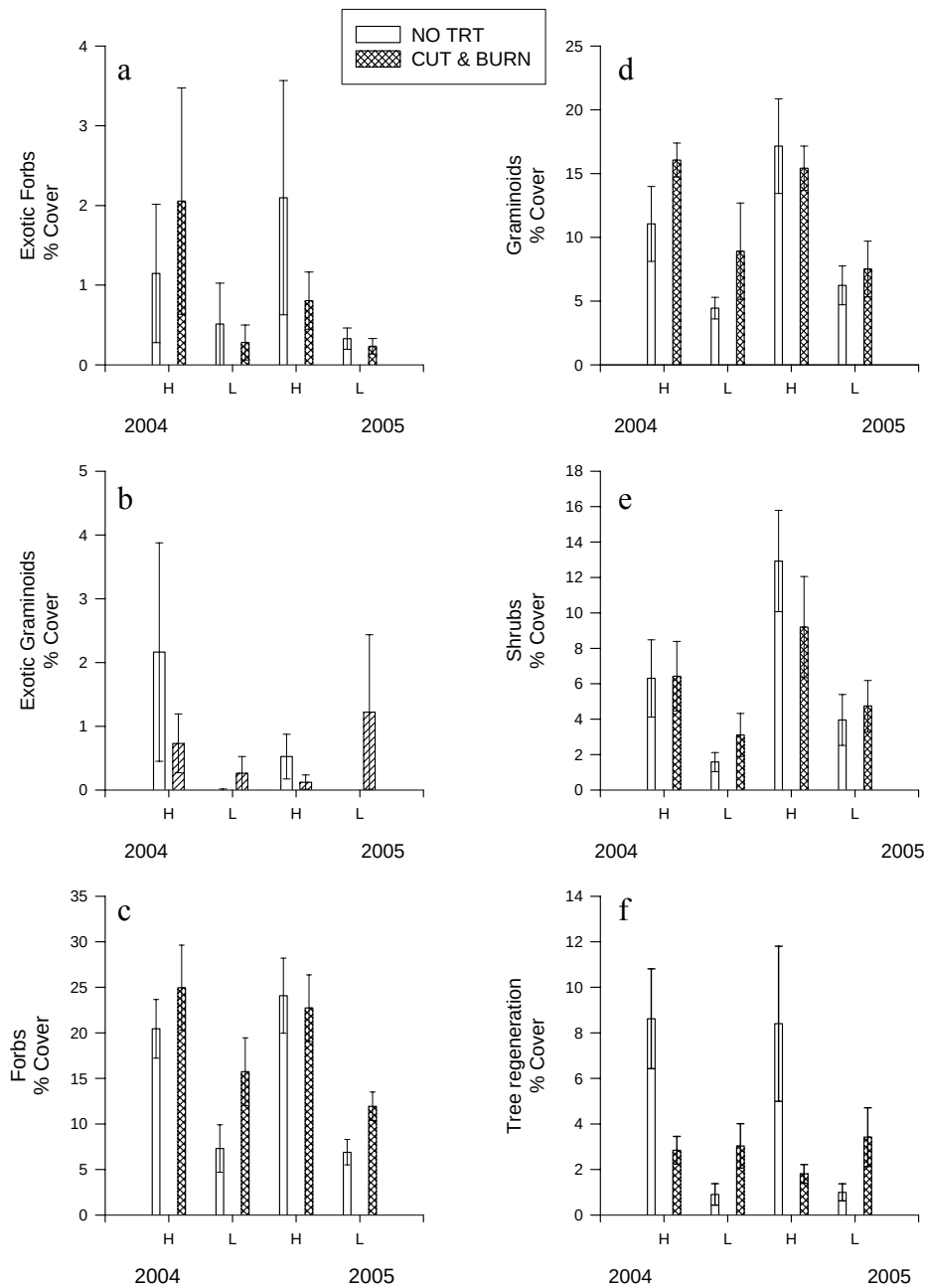


Figure 2. Average (+SE) cover, by plant groups, fire severity (H= high, L= low), treatment, and year. a) Exotic forbs: no significant differences. b) Exotic graminoids: no significant differences. c) All forbs: a significant difference was found between high and low severity in 2004 at $p=.007$ and in 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 (<1.4m tall): the interaction between fire severity and treatment was significant in 2004 at $p=.0137$ and in 2005 at $p=.0014$.

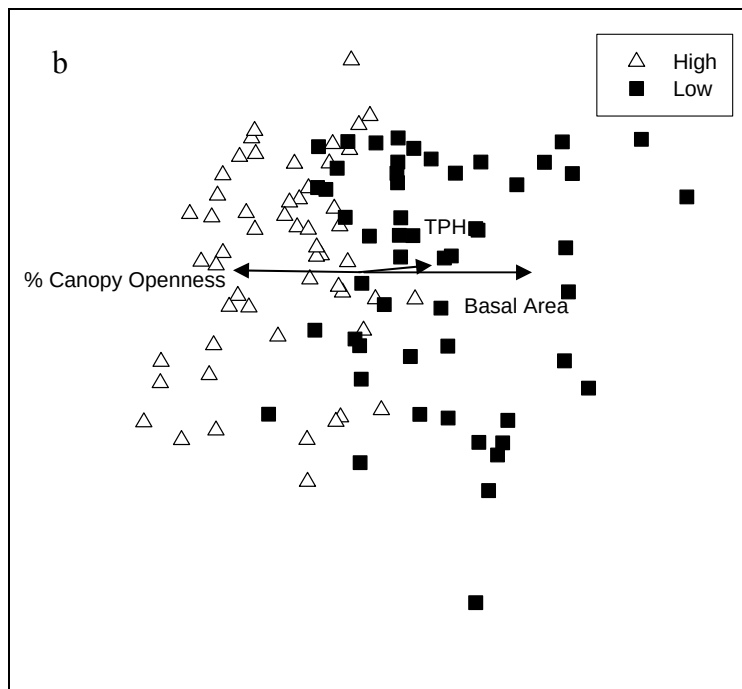
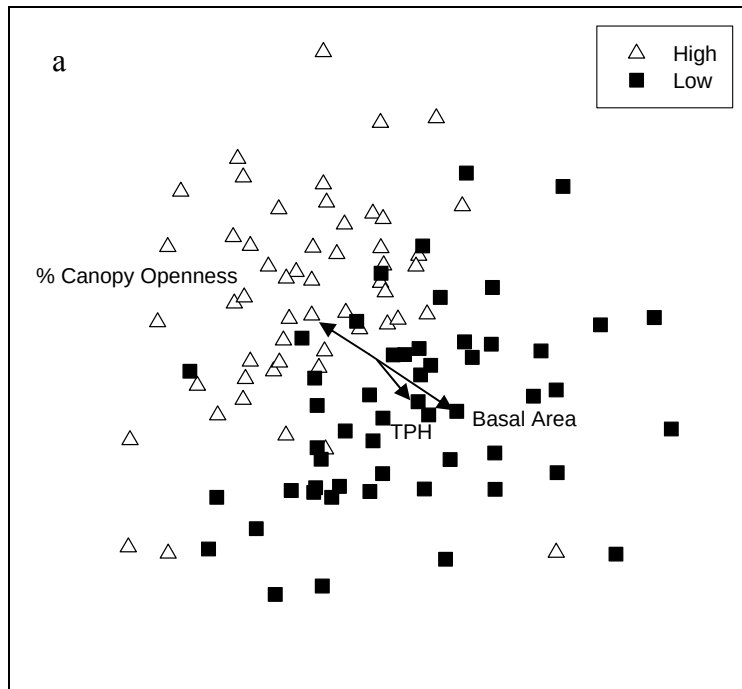


Figure 3. Ordinated plant communities based on plant cover data by severity level. Vector length indicates strength of correlation. a) 2004 data. Stress level is 19.71 for a 3-d solution. Low severity plots were positively correlated with basal area and trees per hectare. b) 2005 data. Stress level is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

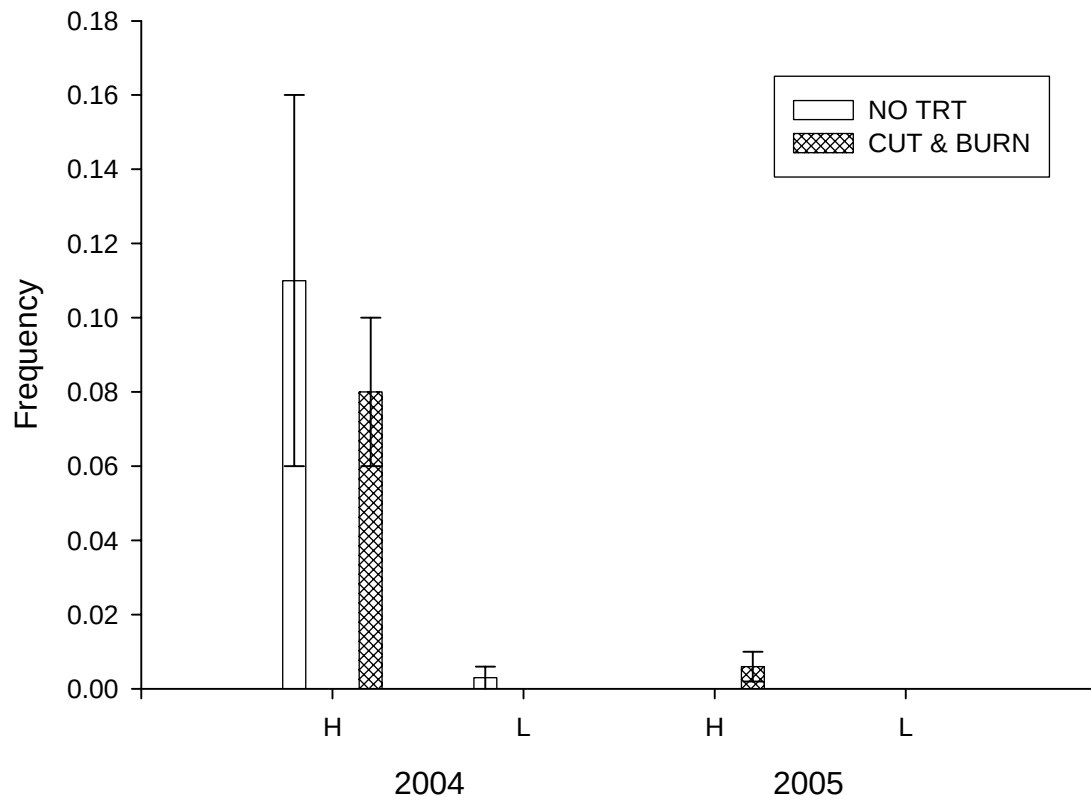


Figure 4. Average frequency (proportion of quadrats + SE) of *Triticum aestivum*, or common wheat, by year, fire severity (H = high, L = low) and treatment.

Chapter 3

Influence of pre-fire fuel reduction treatments on understory response following a severe wildfire, Apache-Sitgreaves National Forests, USA

Abstract

We studied differences in the understory plant communities between paired treated (pre-fire fuel reduction) and untreated sites on the Apache-Sitgreaves National Forests after the Rodeo-Chediski fire in 2004 and 2005, two and three years postfire, respectively. We analyzed species composition and cover using univariate and multivariate methods. We hypothesized that untreated sites would burn more severely and result in more open stands with higher plant cover overall, as well as higher exotic species cover and shrub cover compared to treated stands. Total plant canopy cover in both years and shrub cover in 2005, were higher on untreated areas. No significant differences were found for exotic species cover and overall the cover of exotics was extremely low (<2%). We examined differences in functional groups including C₃ graminoids, C₄ graminoids, legumes, annual/biennial forbs, and perennial forbs. Cover of annual/biennial forbs was significantly higher in untreated areas in both years, due largely to early successional species. Cover of C₃ graminoids and perennial forbs were higher in treated sites in 2005 only. Legume and C₄ graminoid cover did not differ between treated and untreated sites. The fire was seeded with several native and non-native grasses and one penstemon. Using Indicator Species Analyses, we identified several seeded species as well as species common after fires as indicators in untreated areas. It is unclear how long seeded species, as well as other early successional plants,

will persist, but the fire-caused tree mortality in untreated areas is expected to have long-term effects on plant communities.

Introduction

Wildfires in the western United States have increased in size, duration, and frequency over the past several decades (Westerling *et al.* 2006). Although data are limited, fuel reduction treatments that include thinning and slash management have the potential to reduce extreme fire behavior compared to untreated stands (Finney *et al.* 2005, Strom 2005). Strom (2005) found a considerable difference between treated and untreated areas on the Rodeo-Chediski fire. Over 60% of the area that had been thinned and prescribed burned within a decade of the fire burned with low severity, as opposed to only 20% of the area that had not been treated for the same time period. Fuel reduction treatments may also influence post-wildfire understory plant communities, but few studies have attempted to assess the role of pre-fire fuel reduction treatments on understory composition compared to areas that did not receive fuel reduction treatments. The potential role of pre-wildfire fuel reduction on understory communities is dependent on a number of factors, such as the resultant fire severity, the types and abundance of various plant propagules available for colonization, and weather.

Research from a number of wildfires suggests that severely burned areas are more likely to provide habitat for a number of disturbance-tolerant species, increasingly including exotic species. A combination of overstory removal and exposure of bare ground provides habitat for a number of disturbance-dependent species. Griffis *et al.* (2001) compared four treatments in *Pinus ponderosa* P. & C. Lawson var. *scopulorum*

Engelm. (ponderosa pine) forests: unmanaged, thinned, thinned and burned, and wildfire. They found that exotic species richness and abundance increased with disturbance severity; severely burned wildfire sites had more than 40% greater abundance of exotic species than other treatments or unmanaged stands. Crawford *et al.* (2001) found the average cover of exotic species following a 1996 wildfire was twice as high on high severity sites as it was on moderate severity sites. They also found the average cover of *Conyza canadensis* (L.) Cronq. (Canadian horseweed), a disturbance-adapted native, was 27% on moderately burned sites and 18% on high severity sites, as opposed to 0% cover on unburned sites.

The type and abundance of various plant propagules available for post-wildfire colonization also influences understory composition. Response to fire is species-specific (Vose & White, 1991). Some plant species exhibit a strong response after disturbance, including severe wildfire but do not remain in the system over the long term. Foxx (1996) found this relationship for *Chenopodium graveolens* Willd. (fetid goosefoot), which was a dominant species one year after the 1977 La Mesa fire but disappeared by 1985. In addition, fire and fuel treatments can shift the structure of the understory. Species that have the ability to resprout after fire can outcompete species that are obligate seeders (Calvo *et al.*, 2003). Shrubs can often increase after fires by this strategy. *Ceanothus fendleri* Gray (Fendler's ceanothus) increases following burning due to resprouting and fire-stimulated germination (Huffman & Moore 2004).

Previous management practices, including past seeding operations and land use, also affect the plant community after wildfire. Huisinga *et al.* (2005) found greater than 95% native species on burned sites after an intense prescribed fire, and attributed this to a

native-dominated pre-fire plant community due to the absence of a history of thinning and limited grazing in the history of their study site. Thinning and fuel reduction operations also create bare mineral soils and thus potentially increase exotic species (Fulé *et al.* 2005, Wienk *et al.* 2004). However, Hunter *et al.* (2006) found that wildfires had a greater effect on non-native establishment than did fuel treatments. Their study found slightly higher incidence of non-natives in thinned areas at only one of the three wildfires they sampled, but the results were not significant. Exotic species may also establish following severe wildfires due to increased propagule pressure from post-fire seeding practices (Barclay *et al.* 2004, Keeley 2004). Seeding is often done as part of Burned Area Emergency Rehabilitation (BAER) on public lands. However, there is little evidence to suggest that seeding decreases post-fire erosion (Robichaud *et al.* 2000) and seeded species may displace native early successional species (Beyers 2004, Keeley *et al.* 2006).

Finally, extreme variations in precipitation and temperature can greatly affect post-wildfire plant community composition and abundance. Bataineh *et al.* (2006) found a significant difference in species composition 30 years after a wildfire; however they could not differentiate the effects of the fire from drought effects. Moore *et al.* (2006) found that drought was also influential on the understory response following restoration treatments over a 13-year period. They found that while mean standing crop in thinned and burned stands remained much higher than mean standing crop in control stands, it did decrease from 1999-2004, which included one year (2002) in which precipitation was 75% below average. They found C₄ graminoids were not as affected as other functional groups, which they attributed to their greater water-use efficiency. This sensitivity of

understory communities to extreme weather variations in this region complicates our understanding of plant community response to various forest management practices.

The Rodeo-Chediski fire burned 189,000 ha from June to July 2002, creating a fire mosaic across the Apache-Sitgreaves National Forest (A-S) and White Mountain Apache tribal lands, and exhibited some of the most severe fire behavior in Arizona's history. A total of 67,670 ha burned on the A-S National Forest lands (USDA Forest Service 2002), and of those 20,005 ha burned with high severity and were seeded. In addition, 1,325 ha on the east side of the fire were seeded again the following year when more seed supply became available (Kendell Hughes, Apache-Sitgreaves National Forests, personal communication 2006).

The objective of this study was to investigate the effects of pre-wildfire fuel reduction treatments on post-wildfire plant communities on paired, treated and untreated stands. We hypothesized that since untreated plots burned with greater severity and had lower post-fire tree survival compared to treated plots (Strom 2005), total plant canopy cover, cover of disturbance-adapted exotic species, and shrub cover would be higher on untreated plots compared to treated plots.

Methods

Study Area

The Apache-Sitgreaves National Forests lie north of the Mogollon Rim in central-eastern Arizona. Maximum and minimum temperatures, as reported at the Heber Ranger Station by the Western Regional Climate Center (www.wrcc.dri.edu) range from 29.3°C in July down to -8.8°C in January, based on 1950-2005 averages. Average total annual

precipitation is 44.4 cm and average annual total snowfall is 97.3 cm. Precipitation during the years of data collection was lower than the 55-year averages. The area received 33.3 cm of precipitation in 2004 and 40.7 cm in 2005. Total snowfall in winter of 2003/2004 was 50.3 cm and 22.9 cm in winter of 2004/2005. Elevation for the study area ranges from 1,990-2,138 m. Soils are fine, mixed Mollic Eutroboralfs and fine, mixed Typic Eutroboralfs. Forests are dominated by *P. ponderosa*, *Quercus gambelii* Nutt. (Gambel oak), and *Juniperus deppeana* Steud. (alligator juniper)(USDA Forest Service 1989). All sites were above 2000 m to ensure they fell in a *P. ponderosa*-dominated ecosystem, and at less than 45% slope. The average slope was 9%.

Site Selection and Measurement

Our measurements took place from mid-July to early September in 2004 and 2005. We collected data on seven pairs of treated and untreated study sites on the Apache-Sitgreaves National Forests (A-S) that had been established by the USDA Forest Service immediately following the fire in 2002. Non-commercial thinning for fuel reduction had been done between 1990 and 1999. Slash was most commonly pile burned, but on one site slash had been lopped and scattered and on one site slash was crushed. Each treated site was paired with an untreated site of similar topography with no road or other fire break in between (USDA Forest Service 2002, L. Wadleigh and C. Hoffman, pers. comm. 2003, 2006). Data were collected on 2 grids of 3 plots each at each site resulting in 42 plots per treatment (84 plots total).

In 2002, the seed mix, aerially applied at 35.9 kg/ha, contained: *Hordeum vulgare* L. (annual barley), *Lolium perenne* L. (annual rye), *Elymus trachycaulus* (Link) Gould ex Shinnery (slender wheatgrass), and *Bromus marginatus* Nees ex Steud. (*B. marginatus* =

B. carinatus Hook. & Arn.) (Welsh *et al.* 1993) (mountain brome). In 2003, seed was applied again, this time using different seed mixes on the east and west sides of the fire. All seven pairs of sites were on the east side of the fire. The seed mix used on the east side, applied at 23.3 kg/ha, consisted of: *Pascopyrum smithii* (Rydb.) A. Löve (western wheatgrass), *Bouteloua curtipendula* (Michx.) Torr. (sideoats grama), *Festuca ovina* L. (sheep fescue), *Koeleria macrantha* (Ledeb.) J.A. Schultes (prairie Junegrass), *Penstemon palmeri* Gray (Palmer's penstemon), and *H. vulgare*. (Kendell Hughes, Apache-Sitgreaves National Forest, personal communication 2006)

Plot centers were the same as those used in Strom & Fulé (in press), who measured overstory tree structure. At each plot center, we established two 44.8-m transects crossing perpendicularly, resulting in a 31.7-m x 31.7-m square, approximately 1000 m². We measured plant canopy cover in 20- by 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 (Table 1) (Daubenmire 1959). We also estimated cover of litter, rocks, logs, bare ground, and moss. Plant nomenclature and nativity classification follows USDA-NRCS PLANTS Database (2006) and Welsh *et al.* (1993). In addition, we quantified plant species richness by recording all species that occurred within each 1000-m² 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 gathered during the data collection for Strom (2005).

Statistical Methods

We analyzed the data using DISTLM (Anderson 2001 & 2004), a permutation procedure software for multivariate data, so it was an appropriate choice to test differences in the plant community. For consistency, we also used DISTLM for analyzing univariate measures for richness and abundance because these data evidenced non-normality and heterogeneous variances. The univariate tests were run to test for differences in total richness and total abundance (plant canopy cover), plus abundance for the six categories of plant habit. We also followed Moore *et al.* (2006) in analyzing cover by five functional groups: C₃ graminoids, C₄ graminoids, legumes, non-leguminous annual/biennial forbs, and non-leguminous perennial forbs. We used multivariate tests to investigate differences in plant community composition based on presence/absence data and plant canopy cover data for the six plant habit categories. 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 used non-metric multi-dimensional scaling ordination of the cover data by species using PC-ORD software (McCune & Mefford 1999), 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 X percent canopy cover. Species with $p < 0.05$ and an Indicator Value > 25 (Dufrêne & Legendre, 1997) were identified as indicator species.

Results

Based on our reconstruction of pre-fire overstory structure through measurement of living plus fire-killed trees, pre-fire basal area was nearly identical on treated and untreated stands, but tree density was over twice as high on untreated stands (Table 2). There was an 84% decrease in tree density and about a 55% decrease in basal area across treatments after the fire. Tree density of *J. deppeana* decreased by 92%, *P. ponderosa* by more than 60%, and *Q. gambelii* by 42%.

Univariate analysis

We found 334 species on all plots, but our analyses focused only on the 139 species found on at least 5% of the plots. In total, we found 38 exotic species, 13 of which occurred on at least 5% of the plots and therefore were included in the analyses. Total plant species richness did not differ significantly ($p = 0.22$, 2004; $p = 0.72$, 2005) between treated and untreated sites in either year (Figure 1a). However, total plant canopy cover was significantly higher on untreated sites compared to treated sites in both years ($p = 0.002$ in 2004, $p = 0.015$ in 2005) (Figure 1b). Forb cover was significantly higher in both 2004 and 2005 in areas that were not treated compared to untreated areas ($p = 0.009$ in 2004, $p = 0.047$ in 2005). Shrub cover was also higher on untreated sites than on treated sites in 2004 ($p = 0.016$), and was marginally higher on untreated sites in

2005 ($p = 0.054$). Cover of exotic species (forbs or graminoids), graminoids, or regeneration trees did not differ significantly between treated and untreated sites (Figure 2 a-f).

Functional groups

Cover of C_4 (warm season) graminoids did not differ in response to thinning, but cover of C_3 (cool season) graminoids was significantly higher in untreated areas in 2005 ($p = 0.043$) (Figure 3 d, e). Cover of nonleguminous perennial forbs was significantly higher in untreated areas in 2005 ($p = 0.034$). Cover of nonleguminous annual and biennial forbs was significantly higher in untreated areas in both years ($p = 0.026$, 2004 & 2005). Legume cover did not differ significantly on treated and untreated areas in either year (Figure 3 a-c).

Multivariate analysis

Our multivariate analyses indicated that plant community composition, based on presence/absence data, did not differ between treated and untreated areas. However, using species abundance (cover) data, we detected significant differences in the plant community composition between treated and untreated areas ($p = 0.024$ in 2004 and $p = 0.049$ in 2005). The significant differences in plant community composition based on species abundance data were supported by the ordinations of cover data by treatments. A separation can be seen in both years (Figure 4 a,b), and was positively correlated with canopy openness (untreated areas), trees per hectare (TPH) (treated areas) and basal area (treated areas). These ordinations are intended as an illustration of the plant community cover data more so than a quantitative analysis. The Indicator Species Analysis identified 12 indicator species of untreated areas in each year, five of which were indicators in both

years: *C. canadensis*, *B. carinatus*, *Heleomeris multiflora* Nutt. (showy goldeneye), *Cirsium wheeleri* (Gray) Petrak (Wheeler's thistle), and *E. trachycaulus* (Table 3). Only one species, *K. macrantha*, was identified as an indicator species of treated areas.

Discussion

Our hypothesis that the untreated areas, where most trees were killed by fire, would have greater plant cover was upheld. This result matches the results of many other post-fire plant community studies (Crawford *et al.* 2001, Griffis *et al.* 2001, Huisinga *et al.* 2005). The untreated areas burned more severely, as evidenced by higher tree mortality and greater post-fire canopy openness. Sabo (2006) found that basal area must be reduced to $8 \text{ m}^2 \text{ ha}^{-1}$ or less to result in an increase in the understory production. Clary & Ffolliott (1966) found an understory response at $5\text{-}18 \text{ m}^2 \text{ ha}^{-1}$. Basal area in the untreated sites was $2.8 \text{ m}^2 \text{ ha}^{-1}$ after the Rodeo-Chediski fire, but the treated sites, which did not experience as much tree mortality had an average basal area of $13.9 \text{ m}^2 \text{ ha}^{-1}$. Therefore, it is likely that the resulting increase in light on the severely burned untreated sites was responsible to some degree for the increase in plant canopy cover of understory species.

Our hypothesis that there would be greater exotic cover in untreated areas was not upheld. This result contrasts with findings in other managed forests in the Southwest that have experienced severe fire. In the absence of pre-fire vegetation data, we can only assume that this is due largely to an absence of many non-natives in the pre-fire community. However, unlike the Huisinga *et al.* (2005) study that also found low non-native response, our study site does have a history of logging, livestock grazing, and road

building. In addition, many of the typical exotic species found on burned areas were not detected in our study. *Linaria dalmatica* (L.) P. Mill. (Dalmatian toadflax) and *Centaurea diffusa* Lam. (diffuse knapweed) were not detected at all. Dodge (2004) found a strong relationship between fire severity and *L. dalmatica* growth and reproduction. Wolfson *et al.* (2005) reported that *C. diffusa* responds favorably to severe wildfire. *Bromus tectorum* L. (cheatgrass) was observed on only two plots in 2004 and seven plots in 2005 (only once did its cover measure over 1%). *Cirsium vulgare* Savi (Ten) (bull thistle) was observed on only two plots in 2004 and four plots in 2005.

Lastly, the hypothesis that there would be greater shrub cover in untreated areas was upheld in the 2004 season, but not in 2005. This was driven by *C. fendleri*, which was also an indicator species in 2004. It is interesting to note that shrub cover was actually higher in untreated areas in 2005 than it was in untreated areas in 2004. However, there was no significant difference between shrub cover in untreated and treated areas in 2005 due to increased shrub cover in treated areas in 2005 as well.

The analyses by functional groups showed the majority of the species found at higher cover in untreated areas were nonleguminous, annual, biennial, or perennial forbs. This is consistent with what Moore *et al.* (2006) found in restoration treatments, but the response they saw was driven by an exotic species, common mullein. Mullein was not an indicator species in our study. The only exotic species that met the criteria for an indicator was *Lactuca serriola* L. (prickly lettuce), and that was only in 2005. One leguminous species, *L. wrightii*, was identified as an indicator species of untreated plots. Legumes, including a number of *Lotus* species, often increase in abundance after fire (DiTomaso *et al.* 1999).

Moore *et al.*'s (2006) study included 10 years of data and was subject to drought effects. Their study demonstrated that variations in precipitation can affect understory response in this region. C₃ graminoids were dominant until drought occurred. In contrast, our study showed significantly higher cover of C₃ graminoids in 2005, in spite of less winter precipitation that year compared to 2004. C₄ graminoids have better water use efficiency. Moore *et al.* (2006) reported C₄ graminoids withstood the years of drought during their study better than C₃ graminoids. We had lower cover of C₄ graminoids than C₃ graminoids, but if climate trends continue along the trajectory of warming and drought, there may be a shift in the composition of graminoids in the future. Both years of data collection of our study (2004 & 2005) had less-than-average rainfall and were comparable, although there was less snowfall in 2005.

The community composition showed no differences based on presence/absence data, but did show differences based on cover. This was supported by the results of the Indicator Species Analysis. *K. macrantha*, a perennial bunchgrass, was the only indicator species of treated areas. It was also included in the seed mix. Perennial bunch grasses are adapted to low severity fire by sprouting from leaf bases insulated by old leaf sheaths or from below the soil surface where they are protected (Young 1983). The species that were indicators of untreated areas included three other seeded species: *B. carinatus*, *E. trachycaulus*, and *P. smithii*. Many of the remaining indicator species often increase following fire. *C. fendleri* resprouts after fire, and germination is heat-stimulated (Huffman & Moore 2004). Crawford *et al.* (2001) found strong responses to high severity burning for: *C. wheeleri*, *C. canadensis*, *Erigeron divergens* Torr. & Gray (spreading fleabane), *Q. gambelii*, and *Polygonum douglasii* Greene (Douglas' knotweed). *Lotus*

spp. was an indicator in Laughlin et al. (2004) (our study listed *L. wrightii* as an indicator), as was *P. douglasii* (Laughlin et al. 2004). *Penstemon linarioides* Gray (toadflax penstemon) could not be found in the literature as an indicator in other studies. However, *P. barbatus* (Cav.) Roth (beardlip penstemon) has been shown to have a strong germination response to heat and smoke (Abella 2006). *L. serriola* has been shown to increase with high severity fire (Wienk et al. 2004). *H. multiflora*, *Eriogonum racemosum* Nutt. (redroot buckwheat), *B. dissecta* (Gray) Britt., and *Erigeron colomexicanus* A. Nels. (running fleabane) were also indicators of untreated areas.

Conclusions

While this study documented plant community response in the short-term (2- and 3- years postfire), the trends we observed may not persist over time. It is likely that early successional species such as like *C. canadensis* will not persist in the long-term. Some of the seeded species may not persist either, as was seen in long-term monitoring after the 1977 La Mesa fire (Foxy 1996). Strom and Fulé (in press) applied a forest simulation model to the post-Rodeo-Chediski tree structure; simulation results indicated that untreated areas were more likely to undergo a vegetation type conversion from a *P. ponderosa*-dominated ecosystem to a combination of *Quercus* spp., *Juniperus* spp., *Pinus* spp., and *Robinia neomexicana* Gray (New Mexico locust). These changes have long-term implications for the understory plant community. It is likely that high plant cover will persist given a shorter, more open overstory canopy with more deciduous species. Warm-season C₄ graminoids may increase in response to the more open canopy. In general, in areas of high fire severity, such as our untreated sites, there may be a long-

term shift away from shade-tolerant species typically found in the *P. ponderosa* understory. Such sites, with high or complete canopy mortality, make up the majority of the post-fire landscape. Finally, extreme climatic variability, common in this region, will likely play a pivotal role in determining the structure and composition of the post-fire plant community.

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Table 1. Cover classes and midpoints used to classify percent cover within a 20- by 50-cm quadrat (modified from Daubenmire 1959).

Cover Class	Range of Coverage	Midpoint
T	0 – .999 %	.5
1	1 – 5 %	3
2	6 – 25 %	15.5
3	26 – 50 %	38
4	51 – 75 %	63
5	76 – 95 %	85.5
6	96 – 100 %	98

Table 2. Pre- and post-fire overstory characteristics, on treated and untreated sites. Pre-fire values were estimated after the fire by combining living and fire-killed tree data.

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
Untreated	60.1 (1.2)	35.1 (10.1)	1562.7 (620.6)	2.8 (0.8)	18.2 (1.3)
Treated	50.4 (1.2)	321.9 (44.1)	642.7 (134.1)	13.9 (1.2)	18.4 (1.2)

Table 3. Indicator Species¹ of treated and untreated areas on the Apache-Sitgreaves National Forest (table structure modified from Huisinga *et al.*, 2005).

Year	Indicator species	Functional group ²	Seeded species	Indicator value	p-value	Untreated	Treated
2004	Untreated						
	<i>Polygonum douglasii</i>	A		58.0	0.023	2.7 (0.5)	1.0 (0.48)
	<i>Lotus wrightii</i>	L,P		52.3	0.007	1.5 (0.4)	0.5 (0.15)
	<i>Conyza canadensis</i>	A		50.0	0.001	1.0 (0.2)	0.3 (0.12)
	<i>Erigeron divergens</i>	A		46.9	0.003	1.1 (0.3)	0.2 (0.06)
	<i>Eriogonum racemosum</i>	P		46.1	0.005	0.6 (0.1)	0.2 (0.05)
	<i>Quercus gambelii</i>	T		42.5	0.012	1.5 (0.3)	0.6 (0.25)
	<i>Bromus carinatus</i>	C ₃ G	X	41.0	0.002	1.4 (0.4)	0.2 (0.08)
	<i>Heliomeris multiflora</i>	P		38.9	0.003	1.1 (0.5)	0.1 (0.04)
	<i>Cirsium wheeleri</i>	P		30.9	0.036	0.7 (0.2)	0.2 (0.09)
	<i>Erigeron colomexicanus</i>	A		28.7	0.001	0.6 (0.2)	0.1 (0.04)
	<i>Elymus trachycaulus</i>	C ₃ G	X	26.3	0.003	0.8 (0.2)	0.4 (0.15)
	<i>Ceanothus fendleri</i>	S		25.9	0.006	0.6 (0.2)	0.1 (0.05)
	Treated						
	<i>Koeleria macrantha</i>	C ₃ G	X	38.3	0.001	0.1 (0.06)	0.6 (0.16)
2005	Untreated						
	<i>Bahia dissecta</i>	A,P		63.9	0.016	2.4 (0.46)	0.9 (0.22)
	<i>Conyza canadensis</i>	A		61.6	0.001	2.6 (0.45)	1.1 (0.37)
	<i>Heliomeris multiflora</i>	P		57.5	0.001	1.7 (0.60)	0.2 (0.11)
	<i>Erigeron divergens</i>	A		55.1	0.019	1.7 (0.38)	0.5 (0.12)
	<i>Bromus carinatus</i>	C ₃ G	X	47.2	0.001	1.8 (0.40)	0.02(0.02)
	<i>Quercus gambelii</i>	T		42.5	0.012	2.1 (0.43)	1.0 (0.30)
	<i>Cirsium wheeleri</i>	P		41.6	0.008	1.3 (0.35)	0.3 (0.10)
	<i>Lactuca serriola</i> ³	A		39.8	0.025	0.7 (0.20)	0.3 (0.13)
	<i>Erigeron colomexicanus</i>	A		32.7	0.004	1.3 (0.34)	0.3 (0.16)
	<i>Elymus trachycaulus</i>	C ₃ G	X	31.0	0.001	0.7 (0.24)	0.1 (0.05)
	<i>Penstemon linarioides</i>	P		30.8	0.044	1.8 (0.51)	0.7 (0.28)
	<i>Pascopyrum smithii</i>	C ₃ G	X	28.0	0.035	0.8 (0.25)	0.2 (0.09)
	Treated						
	<i>Koeleria macrantha</i>	C ₃ G	X	46.8	0.013	0.2 (0.04)	0.5 (0.12)

¹Species are ranked by Indicator Value within treatments. Average percent cover is given for each species for each treatment. Bold type indicates the site (untreated or treated) with which a species was most strongly associated.

²Functional group codes: C₃ G = C₃ graminoid, A = nonleguminous annual and biennial forb, P = nonleguminous perennial forb, L= leguminous forb, S = shrub, T = tree.

³Exotic species

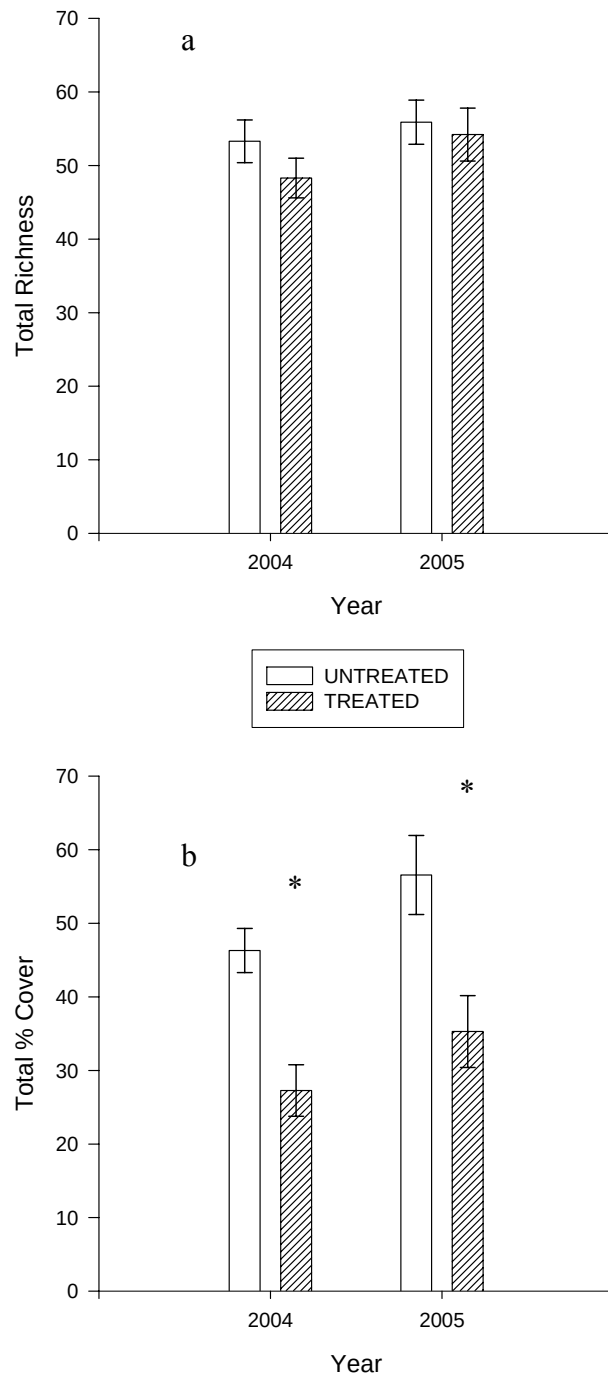


Figure 1. Total richness (no. species/1000m²) (a) and total plant canopy cover (20- by 50-cm quadrats) (b) for treated and untreated sites in 2004 and 2005. An asterisk (*) indicates a significant difference. Error bars are standard errors.

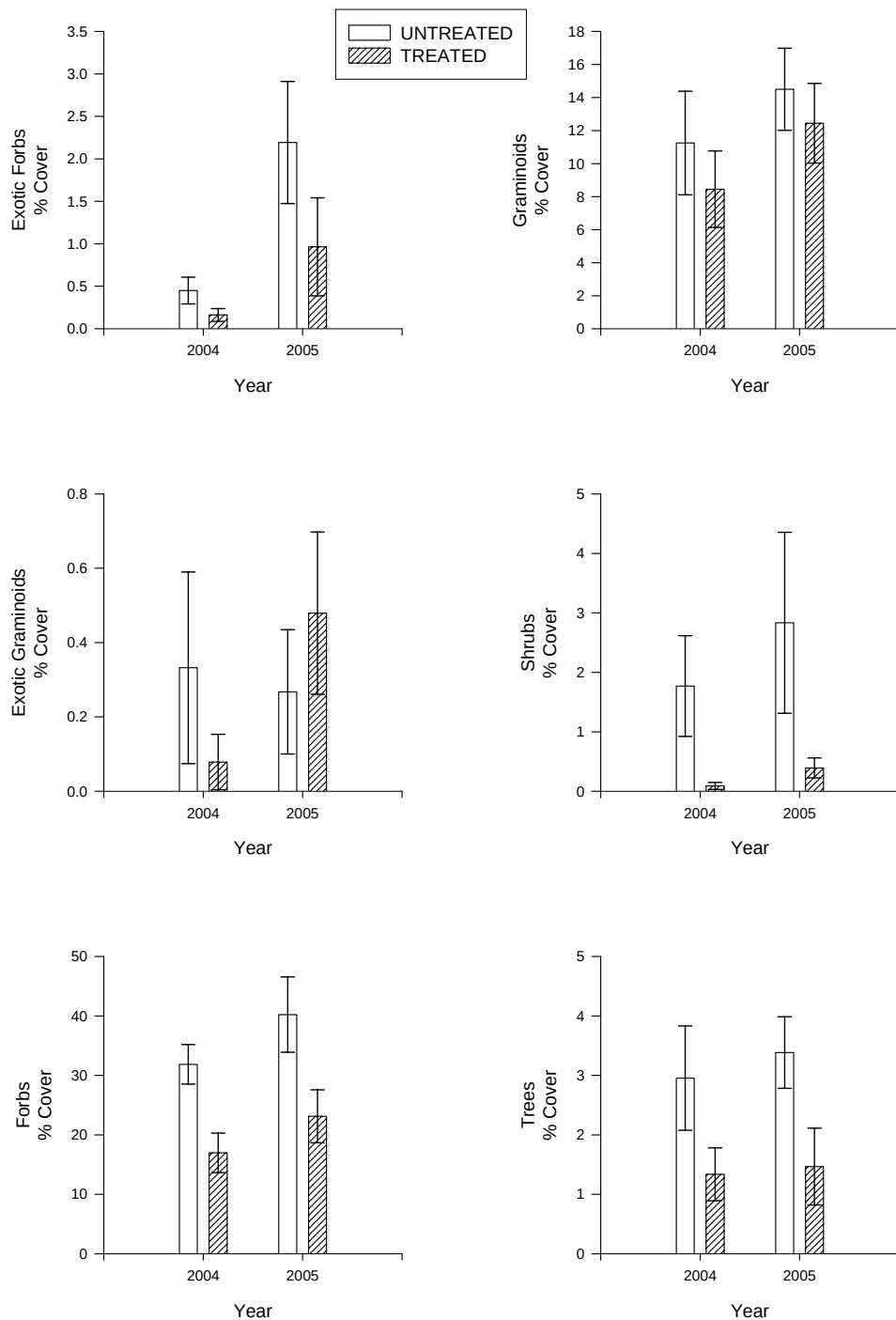


Figure 2. Percent cover (mean + standard error) of a) exotic forbs b) exotic graminoids c) all forbs d) all graminoids e) shrubs and f) tree regeneration < 1.4 m tall for untreated and treated sites in 2004 and 2005.

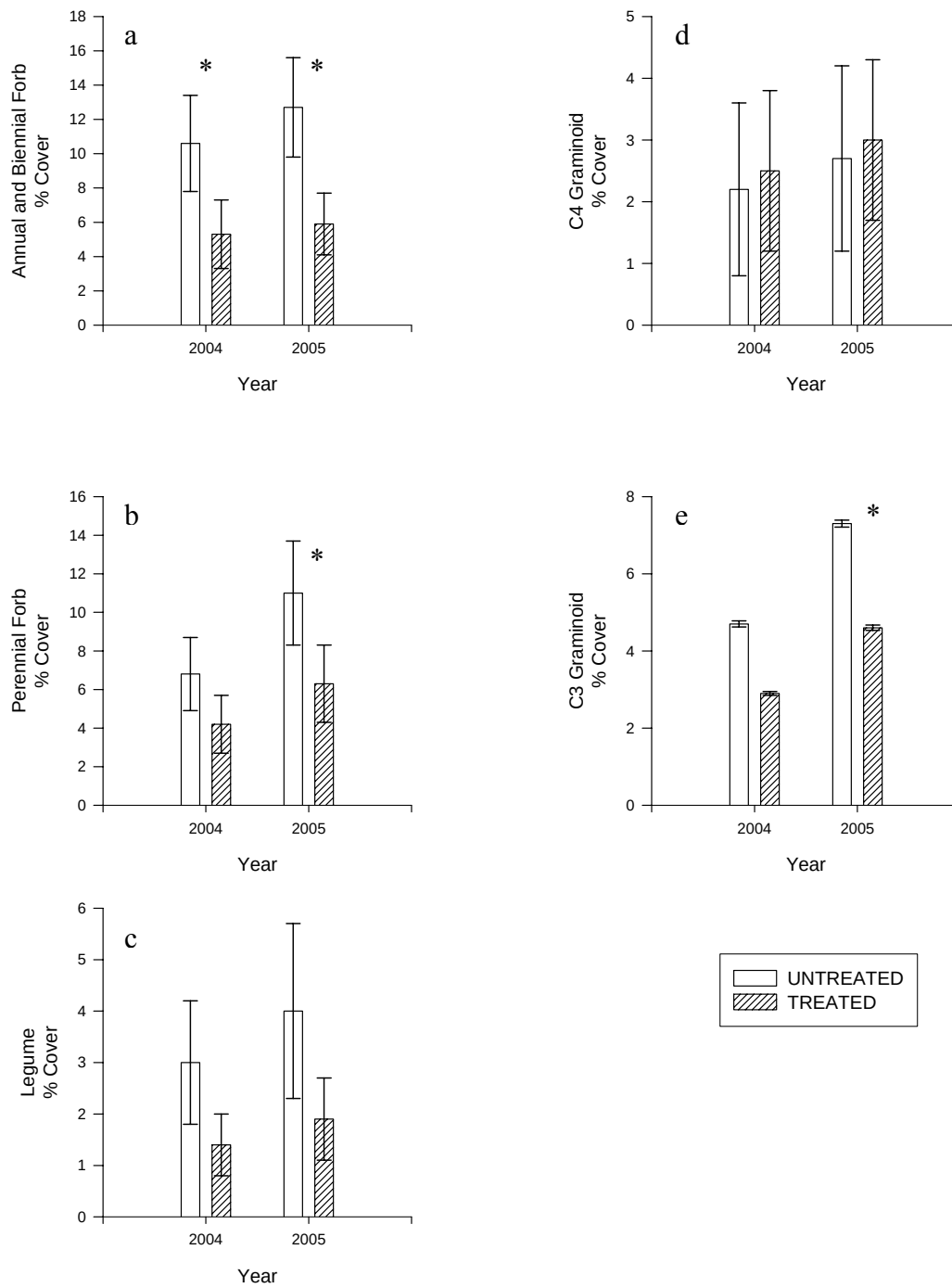


Figure 3. Percent cover (mean + standard error) of a) annual and biennial forbs b) perennial forbs c) legumes d) C₄ graminoids and e) C₃ graminoids for untreated and treated sites in 2004 and 2005.

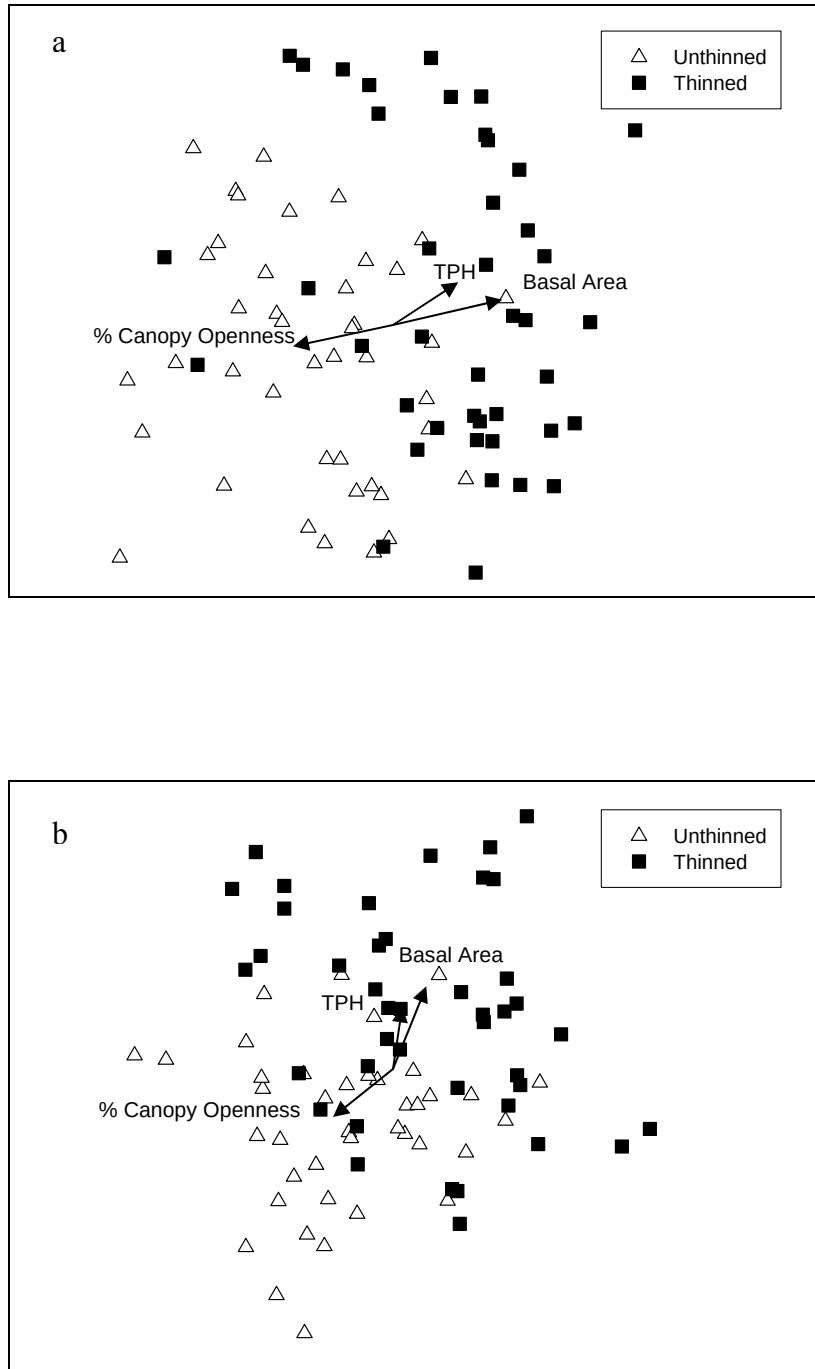


Figure 4. Ordinated plant community cover data by treatment using the Bray-Curtis distance measure. Vector length indicates strength of correlation. a) 2004 data, stress level is 20.33 for a 3-d solution. Treated plots were positively correlated with basal area and trees per hectare. Untreated plots were positively correlated with overstory canopy openness. b) 2005 data, stress level is 20.58 for a 3-d solution. Treated plots were positively correlated with basal area and trees per hectare. Untreated plots were positively correlated with overstory canopy openness.

Chapter 4

Conclusions and Management Implications

Chapters 2 and 3 included discussion of the individual studies. A brief comparison of the two study areas, the White Mountain Apache Tribal lands and the Apache-Sitgreaves National Forests, is presented here along with future research needs and management implications.

Similarities and Differences

Pre-wildfire management practices reduced fire severity on both the White Mountain Apache Tribal lands (WMAT) and the Apache-Sitgreaves National Forests (A-S) (Strom 2005). The main differences in the study designs were that the WMAT study included a severity factor based on the Δ NBR map and the A-S study had a paired (treated/untreated) design. The untreated areas on the A-S were analogous to the high severity sites on the WMAT. Both sites had significantly higher understory plant canopy cover on areas that were not treated, which we attributed to a more open canopy structure caused by higher tree mortality on high severity burns. The ordinated plant communities showed similar relationships between basal area, tree density, and % canopy openness for both studies. Species found in untreated and high severity areas were correlated with higher canopy openness, while low severity and treated sites had higher basal area and tree density. There was no significant difference in plant species richness between treatments on either site. Richness was slightly higher in the WMAT study with 151 species on 5% of plots compared to 138 species on 5% of the plots on the A-S study. There was no significant difference in the amount of exotic species cover in either case. There were 13 exotic species on the WMAT and 14 on the A-S. However, both sites had

plant communities largely dominated by disturbance-adapted native species, based on results from the Indicator Species Analyses.

The seed mixes on each site were composed mostly of native grasses and some native forbs as well. Several of the seeded grass species were indicator species of the high severity/untreated areas (WMAT/A-S respectively), including *Bromus carinatus* Hook. & Arn. (mountain brome), *Elymus trachycaulus* (Link) Gould ex Shinnars (slender wheatgrass), and *Pascopyrum smithii* (Rydb.) A. Löve (western wheatgrass). On both sites, *Koeleria macrantha* (Ledeb.) J.A. Schultes (prairie Junegrass) was an indicator of low severity/treated areas (WMAT/A-S respectively; it was seeded on A-S). The Indicator Species Analysis showed similarities between the plant communities on both sites. Species that were not seeded, but were indicators of high severity/untreated areas included: *Ceanothus fendleri* Gray (Fendler's ceanothus), *Cirsium wheeleri* (Gray) Petrak (Wheeler's thistle), *Conyza canadensis* (L.) Cronq. (Canadian horseweed), *Lotus wrightii* (Gray) Greene (Wright's deervetch), and *Bahia dissecta* (Gray) Britt. (ragleaf bahia) (2005 only). *C. fendleri* resprouts vigorously after fire, and dormant seeds in the seed bank are stimulated to germinate by heat (Huffman & Moore, 2004). *C. wheeleri*, *C. canadensis*, and *B. dissecta* are disturbance tolerant or dependent on disturbances for establishment. *L. wrightii* was shown to be a dominant species on the La Mesa fire by Foxx (1996) and DiTomaso et al. (1999) reported that *Lotus* spp. increased in response to fire. The similarities in plant communities were also due to their proximity to one another.

Management Implications

Given the limited response of exotic species on both studies, these results do not have much bearing on managing for exotics. Maintaining an understory with limited or no exotics is the best prevention for post-wildfire exotic invasion. Using seed mixes for rehabilitation that include only native species is also important for reducing the potential for large-scale invasion. However, any seeding practices, whether native or exotic species are used, has the potential to introduce a suite of non-seedmix species. The limited response of exotic species is unexpected, as that these lands have been actively managed for all of their recent history. The A-S is public land, which is used for recreation, timber, and livestock grazing, and has been seeded with exotic species for decades. The WMAT lands have been managed in a similar manner, but have less public access for recreational purposes. In both cases, however, previous management would lead one to expect that a large-scale exotic invasion would occur after a fire of the size and severity of the Rodeo-Chediski.

These two companion studies offer insight to the short-term response of understory vegetation to a large, severe wildfire. In the future, a long-term portrait of the understory will be more useful for determining post-fire management of the understory as was done in Foxx (1996). Future data collection is needed to assess the persistence of seeded species and early successional plants. In addition, the exotics currently present in low cover do have the potential to establish. *Bromus tectorum* L. (cheatgrass) and *Cirsium vulgare* (Savi) Ten. (bull thistle) are both currently present in very low cover, but whether they become established and spread remains to be seen. Other noxious species such as *Convolvulus arvensis* L. (field bindweed) and *Onopordum acanthium* L. (Scotch

thistle), which were found on less than 5% of plots on the WMAT, also have the potential to establish and spread over a larger area.

Overall, the most important lesson learned from the Rodeo-Chediski fire is that fuel reduction treatments, including prescribed burning and thinning, result in lower severity burns (Strom 2005). The study design on the A-S examined forest structure at the stand level. The WMAT study was designed over a larger landscape scale, but the two studies show analogous results. If wildfires do continue to increase as projected with global warming trends (Westerling *et al.* 2006), larger expanses of understory will become affected. In addition, if the modeling scenarios in Strom & Fulé (in press) do come to fruition at a landscape scale, this will have broad implications for the understory. As ecosystems shift from *Pinus ponderosa* P. & C. Lawson (ponderosa pine) dominated to shrubfield dominated systems, typical shade-tolerant species may be replaced by other functional groups, such as C₄ graminoids. Given the current state of southwestern forests, this could occur on a vast scale. This creates a conundrum for land managers. Fuel reduction treatments are necessary for reducing high severity fire, but at the same time such treatments may create disturbances that support a number of invasive exotic species (Sieg *et al.* 2003). Careful monitoring of exotic species populations and planning to avoid their spread are needed in concert with fuel reduction treatments.

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APPENDICES

Appendix i. Species list for White Mountain Apache Tribal lands study

Species name	Authority	Origin	Growth Habit
<i>Achillea millefolium</i>	L.		Forb
<i>Achnatherum nelsonii</i>	(Scribn.) Barkworth		Graminoid
<i>Agropyron desertorum</i>	(Fish. Ex Link) J.A. Schultes	Exotic	Graminoid
<i>Amaranthus hybridus</i>	L.		Forb
<i>Amaranthus powellii</i>	S. Wats.		Forb
<i>Ambrosia acanthicarpa</i>	Hook.		Forb
<i>Antennaria marginata</i>	Greene		Forb
<i>Antennaria parvifolia</i>	Nutt.		Forb
<i>Apocynum androsaemifolium</i>	L.		Shrub
<i>Arabis</i> sp.			Forb
<i>Arctostaphylos pringlei</i>	Parry		Shrub
<i>Arctostaphylos pungens</i>	Kunth		Shrub
<i>Arenaria lanuginosa</i>	(Michx.) Rohrb.		Forb
<i>Arenaria lanuginosa</i> ssp. <i>saxosa</i>	(Gray) Maguire		Forb
<i>Artemisia carruthii</i>	Wood ex Carruth.		Forb
<i>Artemisia ludoviciana</i>	(Nutt.) Keck		Forb
<i>Astragalus humistratus</i>	Gray		Forb
<i>Astragalus</i> sp.			Forb
<i>Astragalus tephrodes</i>	Gray		Forb
<i>Bahia dissecta</i>	(Gray) Britt.		Forb
<i>Blepharoneuron tricholepis</i>	(Torr.) Nash		Graminoid
<i>Bouteloua curtipendula</i>	(Michx.) Torr.		Graminoid
<i>Bouteloua gracilis</i>	(Willd. ex Kunth) Lag. ex Griffiths		Graminoid
<i>Brickellia grandiflora</i>	(Hook.) Nutt.		Forb
<i>Bromus carinatus</i>	Hook. & Arn.		Graminoid
<i>Bromus ciliatus</i>	L.		Graminoid
<i>Bromus japonicus</i>	Thunb. ex Murr.	Exotic	Graminoid
<i>Bromus tectorum</i>	L.	Exotic	Graminoid
<i>Calliandra humilis</i>	Benth.		Forb
<i>Carex</i> sp.			Graminoid
<i>Carex geophila</i>	Mack.		Graminoid
<i>Castilleja</i> sp.			Forb
<i>Ceanothus fendleri</i>	Gray		Shrub
<i>Cercocarpus montanus</i>	Raf.		Shrub
<i>Chaetopappa ericoides</i>	(Torr.) Nesom		Forb
<i>Chamaesyce fendleri</i>	(Torr. & Gray) Small		Forb
<i>Chenopodium album</i>	L.	Exotic	Forb
<i>Chenopodium graveolens</i>	Willd.		Forb
<i>Chenopodium</i> sp.	L.		Forb
<i>Cirsium vulgare</i>	(Savi) Ten	Exotic	Forb
<i>Cirsium wheeleri</i>	(Gray) Petrak		Forb
<i>Cologania angustifolia</i>	Kunth		Forb
<i>Commelina dianthifolia</i>	Delile		Forb
<i>Comandra umbellata</i>	(L.) Nutt.		Forb
<i>Conyza canadensis</i>	(L.) Cronq.		Forb
<i>Coreopsis tinctoria</i>	Nutt.		Forb
<i>Cordylanthus wrightii</i>	Gray		Forb
<i>Cyperus</i> sp.			Graminoid
<i>Cyperus fendlerianus</i>	Boeckl.		Graminoid
<i>Dactylis glomerata</i>	L.	Exotic	Graminoid
<i>Dalea albiflora</i>	Gray		Forb

Species name	Authority	Origin	Growth Habit
<i>Dalea candida</i>	Willd.		Forb
<i>Desmodium grahamii</i>	Gray		Forb
<i>Dyssodia papposa</i>	(Vent.) A.S. Hitchc.		Forb
<i>Elymus elymoides</i>	(Raf.) Swezey		Graminoid
<i>Elymus trachycaulus</i>	(Link) Gould ex Shinners		Graminoid
<i>Epilobium brachycarpum</i>	K. Presl.		Forb
<i>Eriogonum alatum</i>	Torr.		Forb
<i>Erigeron colomexicanus</i>	A. Nels.		Forb
<i>Erigeron divergens</i>	Torr. & Gray		Forb
<i>Erigeron flagellaris</i>	Gray		Forb
<i>Erigeron neomexicanus</i>	Gray		Forb
<i>Eriogonum racemosum</i>	Nutt.		Forb
<i>Erigeron speciosus</i>	(Lindl.) DC.		Forb
<i>Frasera speciosa</i>	Dougl. ex Griseb.		Forb
<i>Geranium caespitosum</i>	James		Forb
<i>Hedeoma hyssopifolia</i>	Gray		Forb
<i>Heliomeris multiflora</i>	Nutt.		Forb
<i>Heterotheca villosa</i>	(Pursh) Shinners		Forb
<i>Hieracium fendleri</i>	Schultz-Bip.		Forb
<i>Houstonia wrightii</i>	Gray		Forb
<i>Hymenoxys bigelovii</i>	(Gray) Parker		Forb
<i>Hymenopappus filifolius</i>	Hook.		Forb
<i>Hymenopappus mexicanus</i>	Gray		Forb
<i>Ipomopsis aggregata</i>	(Pursh) V. Grant		Forb
<i>Ipomoea costellata</i>	Torr.		Forb
<i>Juniperus deppeana</i>	Steud.		Tree
<i>Koeleria macrantha</i>	(Ledeb.) J.A. Schultes		Graminoid
<i>Lactuca serriola</i>	L.	Exotic	Forb
<i>Lathyrus laetivirens</i>	Greene ex Rydb.		Forb
<i>Lathyrus lanszwertii</i> var. <i>leucanthus</i>	(Rydb.) Dorn		Forb
<i>Lepidium densiflorum</i>	Schrad.		Forb
<i>Linum aristatum</i>	Engelm.		Forb
<i>Linum lewisii</i> var. <i>lewisii</i>	Pursh		Forb
<i>Linum neomexicanum</i>	Greene		Forb
<i>Lithospermum incisum</i>	Lehm.		Forb
<i>Lithospermum multiflorum</i>	Torr. ex Gray		Forb
<i>Lotus wrightii</i>	(Gray) Greene		Forb
<i>Lupinus argenteus</i>	Pursh		Forb
<i>Machaeranthera canescens</i>	(Pursh) Gray		Forb
<i>Mahonia repens</i>	(Lindl.) G. Don		Shrub
<i>Medicago lupulina</i>	L.	Exotic	Forb
<i>Mirabilis decipiens</i>	(Standl.) Standl.		Forb
<i>Muhlenbergia longiligula</i>	A.S. Hitchc.		Graminoid
<i>Muhlenbergia minutissima</i>	(Steud.) Swallen		Graminoid
<i>Muhlenbergia montana</i>	(Nutt.) A.S. Hitchc.		Graminoid
<i>Muhlenbergia virescens</i>	(Kunth) Kunth		Graminoid
<i>Nassella viridula</i>	(Trin.) Barkworth		Graminoid
<i>Oenothera elata</i>	Kunth'		Forb
<i>Oenothera pubescens</i>	Willd. ex Spreng.		Forb
<i>Opuntia</i> sp.			Cactus
<i>Packera neomexicana</i>	(Gray) W.A. Weber & A. Love		Forb
<i>Panicum bulbosum</i>	Kunth		Graminoid
<i>Pascopyrum smithii</i>	(Rydb.) A. Love		Graminoid
<i>Pedicularis centranthera</i>	Gray		Forb
<i>Penstemon barbatus</i>	(Cav.) Roth		Forb

Species name	Authority	Origin	Growth Habit
<i>Penstemon linarioides</i>	Gray		Forb
<i>Pennellia longifolia</i>	(Benth.) Rollins		Forb
<i>Penstemon virgatus</i>	Gray		Forb
<i>Phaseolus angustissimus</i>	Gray		Forb
<i>Phaseolus grayanus</i>	Woot. & Standl.		Forb
<i>Phlox gracilis</i>	(Hook.) Greene		Forb
<i>Pinus ponderosa</i>	P. & C. Lawson		Tree
<i>Piptochaetium pringlei</i>	(Beal) Parodi		Graminoid
<i>Poa fendleriana</i>	(Steud.) Vasey		Graminoid
<i>Polygonum aviculare</i>	L.	Exotic	Forb
<i>Polygonum douglasii</i>	Greene		Forb
<i>Portulaca oleracea</i>	L.		Forb
<i>Pseudognaphalium macounii</i>	(Greene) Kartesz		Forb
<i>Pseudotsuga menziesii</i>	(Mirb.)		Tree
<i>Pseudocymopterus montanus</i>	(Gray) Coult. & Rose		Forb
<i>Quercus gambelii</i>	Nutt.		Tree
<i>Quercus grisea</i>	Liebm.		Shrub
<i>Quercus turbinella</i>	Greene		Shrub
<i>Rhus glabra</i>	L.		Tree
<i>Rhus trilobata</i>	Nutt.		Shrub
<i>Robinia neomexicana</i>	Gray		Tree
<i>Rosa woodsii</i>	Lindl.		Shrub
<i>Rubus idaeus</i>	(Michx.) Focke		Shrub
<i>Rudbeckia hirta</i>	L.		Forb
<i>Schoenocrambe linearifolia</i>	(Gray) Rollins		Forb
<i>Schizachyrium scoparium</i>	(Michx.) Nash		Graminoid
<i>Senecio wootonii</i>	Greene		Forb
<i>Silene antirrhina</i>	L.		Forb
<i>Silene laciniata</i>	Cav.		Forb
<i>Solidago sp.</i>	L.		Forb
<i>Sporobolus compositus</i>	(Poir.) Merr.		Graminoid
<i>Sporobolus cryptandrus</i>	(Torr.) Gray		Graminoid
<i>Sporobolus interruptus</i>	Vasey		Graminoid
<i>Symphyotrichum falcatum</i>	Lindl.		Forb
<i>Taraxacum officinale</i>	G.H.Weber ex Wiggers	Exotic	Forb
<i>Thalictrum fendleri</i>	Engelm. ex Gray		Forb
<i>Thermopsis montana var. montana</i>	Nutt.		Forb
<i>Thlaspi montanum</i>	L.		Forb
<i>Townsendia exscapa</i>	(Richards.) Porter		Forb
<i>Tragopogon dubius</i>	Scop.	Exotic	Forb
<i>Tradescantia pinetorum</i>	Greene		Forb
<i>Triticum aestivum</i>	L.	Exotic	Graminoid
<i>Verbascum thapsus</i>	L.	Exotic	Forb
<i>Vicia pulchella</i>	Kunth		Forb
<i>Wyethia arizonica</i>	Gray		Forb

Appendix ii. Species list for Apache-Sitgreaves National Forests study.

Species name	Authority	Origin	Growth Habit
<i>Achillea millefolium</i>	L.		Forb
<i>Amaranthus hybridus</i>	L.		Forb
<i>Antennaria parvifolia</i>	Nutt.		Forb
<i>Arctostaphylos pringlei</i>	Parry		Shrub
<i>Arctostaphylos pungens</i>	Kunth		Shrub
<i>Arenaria lanuginosa</i>	(Michx.) Rohrb.		Forb
<i>Arenaria lanuginosa ssp. saxosa</i>	(Gray) Maguire		Forb
<i>Aristida arizonica</i>	Vasey		Graminoid
<i>Artemisia carruthii</i>	Wood ex Carruth.		Forb
<i>Artemisia ludoviciana</i>	(Nutt.) Keck		Forb
<i>Astragalus humistratus</i>	Gray		Forb
<i>Astragalus sp.</i>			Forb
<i>Astragalus tephrodes</i>	Gray		Forb
<i>Bahia dissecta</i>	(Gray) Britt.		Forb
<i>Blepharoneuron tricholepis</i>	(Torr.) Nash		Graminoid
<i>Bouteloua curtipendula</i>	(Michx.) Torr.		Graminoid
<i>Bouteloua gracilis</i>	(Willd. ex Kunth) Lag. ex Griffiths		Graminoid
<i>Bromus carinatus</i>	Hook. & Arn.		Graminoid
<i>Bromus inermis</i>	Leyss.	Exotic	Graminoid
<i>Bromus japonicus</i>	Thunb. ex Murr.	Exotic	Graminoid
<i>Bromus tectorum</i>	L.	Exotic	Graminoid
<i>Calliandra humilis</i>	Benth.		Forb
<i>Carex sp.</i>			Graminoid
<i>Carex geophila</i>	Mack.		Graminoid
<i>Castilleja sp.</i>			Forb
<i>Ceanothus fendleri</i>	Gray		Shrub
<i>Chaetopappa ericoides</i>	(Torr.) Nesom		Forb
<i>Chamaesyce fendleri</i>	(Torr. & Gray) Small		Forb
<i>Chenopodium album</i>	L.	Exotic	Forb
<i>Chenopodium graveolens</i>	Willd.		Forb
<i>Chenopodium leptophyllum</i>	(Moq.) Nutt. ex S. Wats.		Forb
<i>Chenopodium sp.</i>	L.		Forb
<i>Cirsium vulgare</i>	(Savi) Ten	Exotic	Forb
<i>Cirsium wheeleri</i>	(Gray) Petrak		Forb
<i>Cologania angustifolia</i>	Kunth		Forb
<i>Commelina dianthifolia</i>	Delile		Forb
<i>Comandra umbellata</i>	(L.) Nutt.		Forb
<i>Conyza canadensis</i>	(L.) Cronq.		Forb
<i>Cyperus fendlerianus</i>	Boeckl.		Graminoid
<i>Dalea albiflora</i>	Gray		Forb
<i>Dalea candida</i>	Willd.		Forb
<i>Desmanthus cooleyi</i>	(Eat.) Trel.		Forb
<i>Dichanthelium oligosanthos</i>	(J. A. Schultes) Gould (Cham. & Schlecht.) Fenzl ex		Graminoid
<i>Drymaria leptophylla</i>	Rohrb.		Forb
<i>Echeandia flavescens</i>	(J.A. & J.H. Schultes) Cruden		Forb
<i>Elymus elymoides</i>	(Raf.) Swezey		Graminoid
<i>Elymus trachycaulus</i>	(Link) Gould ex Shinners		Graminoid
<i>Epilobium brachycarpum</i>	K. Presl.		Forb
<i>Eragrostis pectinacea var. pectinacea</i>	(Michx.) Nees ex Steud		Graminoid
<i>Eriogonum alatum</i>	Torr.		Forb
<i>Erigeron colomexicanus</i>	A. Nels.		Forb

Species name	Authority	Origin	Growth Habit
<i>Erigeron divergens</i>	Torr. & Gray		Forb
<i>Erigeron flagellaris</i>	Gray		Forb
<i>Erigeron neomexicanus</i>	Gray		Forb
<i>Eriogonum racemosum</i>	Nutt.		Forb
<i>Erodium cicutarium</i>	(L.) L'Her. ex Ait.	Exotic	Forb
<i>Gayophytum diffusum</i>	Torr. & Gray		Forb
<i>Geranium caespitosum</i>	James		Forb
<i>Gutierrezia sarothrae</i>	(Pursh) Britt. & Rusby		Shrub
<i>Hedeoma hyssopifolia</i>	Gray		Forb
<i>Helianthus annuus</i>	L.		Forb
<i>Heliomeris multiflora</i>	Nutt.		Forb
<i>Heterotheca villosa</i>	(Pursh) Shinnery		Forb
<i>Hieracium fendleri</i>	Schultz-Bip.		Forb
<i>Hordeum jubatum</i>	L.		Graminoid
<i>Houstonia wrightii</i>	Gray		Forb
<i>Hymenoxys bigelovii</i>	(Gray) Parker		Forb
<i>Hymenopappus filifolius</i>	Hook.		Forb
<i>Hymenopappus mexicanus</i>	Gray		Forb
<i>Hymenothrix wrightii</i>	Gray		Forb
<i>Ipomoea costellata</i>	Torr.		Forb
<i>Ipomopsis multiflora</i>	(Nutt.) V. Grant		Forb
<i>Juniperus deppeana</i>	Steud.		Tree
<i>Koeleria macrantha</i>	(Ledeb.) J.A. Schultes		Graminoid
<i>Lactuca serriola</i>	L.	Exotic	Forb
<i>Lactuca tatarica</i>	(L.) C.A. Mey.		Forb
<i>Lathyrus laetivirens</i>	Greene ex Rydb.		Forb
<i>Lepidium densiflorum</i>	Schrad.		Forb
<i>Linum neomexicanum</i>	Greene		Forb
<i>Lolium perenne ssp. perenne</i>	L.	Exotic	Graminoid
<i>Lotus wrightii</i>	(Gray) Greene		Forb
<i>Lupinus argenteus</i>	Pursh		Forb
<i>Machaeranthera gracilis</i>	(Nutt.) Shinnery		Forb
<i>Mirabilis decipiens</i>	(Standl.) Standl.		Forb
<i>Muhlenbergia longiligula</i>	A.S. Hitchc.		Graminoid
<i>Muhlenbergia minutissima</i>	(Steud.) Swallen		Graminoid
<i>Muhlenbergia montana</i>	(Nutt.) A.S. Hitchc.		Graminoid
<i>Muhlenbergia virescens</i>	(Kunth) Kunth		Graminoid
<i>Muhlenbergia wrightii</i>	Vasey ex Coult.		Graminoid
<i>Nicotiana attenuata</i>	Torr. ex S. Wats.		Forb
<i>Oenothera pubescens</i>	Willd. ex Spreng.		Forb
<i>Opuntia sp.</i>			Cactus
<i>Packera neomexicana</i>	(Gray) W.A. Weber & A. Love		Forb
<i>Pascopyrum smithii</i>	(Rydb.) A. Love		Graminoid
<i>Pedicularis centranthera</i>	Gray		Forb
<i>Penstemon barbatus</i>	(Cav.) Roth		Forb
<i>Penstemon linarioides</i>	Gray		Forb
<i>Pennellia longifolia</i>	(Benth.) Rollins		Forb
<i>Penstemon virgatus</i>	Gray		Forb
<i>Phaseolus angustissimus</i>	Gray		Forb
<i>Phlox gracilis</i>	(Hook.) Greene		Forb
<i>Phlox longifolia</i>	Nutt.		Forb
<i>Pinus ponderosa</i>	P. & C. Lawson		Tree
<i>Plantago argyrea</i>	Morris		Forb
<i>Poa compressa</i>	L.	Exotic	Graminoid
<i>Poa fendleriana</i>	(Steud.) Vasey		Graminoid

Species name	Authority	Origin	Growth Habit
<i>Poa pratensis</i>	L.	Exotic	Graminoid
<i>Polygonum douglasii</i>	Greene		Forb
<i>Portulaca oleracea</i>	L.		Forb
<i>Pseudognaphalium macounii</i>	(Greene) Kartesz		Forb
<i>Pseudocymopterus montanus</i>	(Gray) Coult. & Rose		Forb
<i>Psoralidium lanceolatum</i>	(Pursh.) Rydb.		Forb
<i>Psoralidium tenuiflorum</i>	(Pursh.) Rydb.		Forb
<i>Quercus gambelii</i>	Nutt.		Tree
<i>Quercus grisea</i>	Liebm.		Shrub
<i>Quercus turbinella</i>	Greene		Shrub
<i>Robinia neomexicana</i>	Gray		Tree
<i>Schoenocrambe linearifolia</i>	(Gray) Rollins		Forb
<i>Schizachyrium scoparium</i>	(Michx.) Nash		Graminoid
<i>Senecio flaccidus</i>	Less.		Forb
<i>Silene antirrhina</i>	L.		Forb
<i>Silene laciniata</i>	Cav.		Forb
<i>Solidago sp.</i>	L.		Forb
<i>Sporobolus cryptandrus</i>	(Torr.) Gray		Graminoid
<i>Sporobolus interruptus</i>	Vasey		Graminoid
<i>Symphyotrichum falcatum</i>	Lindl.		Forb
<i>Taraxacum officinale</i>	G.H. Weber ex Wiggers	Exotic	Forb
<i>Thelesperma megapotamicum</i>	(Spreng.) Kuntze		Forb
<i>Thlaspi montanum</i>	L.		Forb
<i>Townsendia exscapa</i>	(Richards.) Porter		Forb
<i>Tragopogon dubius</i>	Scop.	Exotic	Forb
<i>Tradescantia pinetorum</i>	Greene		Forb
<i>Triticum aestivum</i>	L.	Exotic	Graminoid
<i>Verbascum thapsus</i>	L.	Exotic	Forb
<i>Vicia americana</i>	Muhl. ex Willd.		Forb
<i>Vulpia octoflora</i>	(Walt.) Rydb.		Graminoid
<i>Wyethia arizonica</i>	Gray		Forb
<i>Yucca angustissima</i>	Engelm. ex Trel.		Shrub