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RESEARCH ARTICLE

Post-Fire Control of Invasive Plants Promotes Native Recovery in a Burned Desert Shrubland

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Abstract

Invasive annual grasses have become increasingly important components of desert vegetation in North America. They are especially problematic because they increase the extent, severity, and frequency of fire in desert shrublands that normally experience fire very rarely, or not at all. After fire, invasive grasses and forbs are often dominant and restoration methods are required to promote native plant recovery. Three treatments to control invasive annual grasses and forbs were implemented in the first 3 years following a fire in creosote bush scrub vegetation. Treatments included early season mechanical removal (raking) of all annuals, grass-specific herbicide (Fusilade II), and Fusilade II plus hand pulling of exotic forbs. In the first year, all treatments reduced invasive annual grass abundance by about half but had little effect on native annuals. Treatment effectiveness was minimal in the first year due to low and irregular distribution of rainfall. In the second year, insufficient rainfall prevented the germination of any annual plants and no treatments were applied. In the third year, precipitation onset occurred later in the season and was above average. Although the raking treatment performed poorly, treatments utilizing Fusilade II nearly eliminated invasive grasses and forbs, achieved native annual dominance, and increased native perennial abundance. These results indicate that in the absence of invasive grasses and forbs, the native annual community can be resilient to fire disturbance and native perennials can recover. The results also suggest that burned creosote bush shrublands can be managed after fire to decrease the chance of invasive plant–fire feedback.

Key words: **California, creosote bush scrub, fire feedback, Fusilade, grass/fire cycle, restoration.**

Introduction

Fires have historically been rare in creosote bush scrub vegetation, but are now occurring with increased frequency in the Mojave and Colorado Deserts of California, U.S.A., fueled by invasive annual grasses (Brooks & Esque 2002). Fire in this ecosystem can be problematic for a number of reasons. First, these low productivity shrublands have long-lived dominant shrubs (Vasek 1980; Bowers et al. 1995) with slow population dynamics and low turnover compared with the lifespan of humans (Cody 2000). Recovery after fire could take at least 100 years depending on various factors, but probably much longer (Lovich & Bainbridge 1999). Second, most of the native shrub species lack traits that aid their recovery after fire and can be locally extirpated in worst-case scenarios (Brown & Minnich 1986; Brooks & Minnich 2006). Third, invasive annual grasses appear to be less affected by fire than native species and often dominate the post-fire vegetation (Brooks & Esque 2002). Fourth, experimental evidence suggests that invasive annual grasses competitively suppress native perennial (DeFalco et al. 2007) and annual species (Brooks 2000; DeFalco et al. 2003; Steers 2008). Finally, if invasive grasses dominate post-burn vegetation, they also increase the likelihood that the site will burn again in a process known as the invasive plant–fire regime cycle (Brooks et al. 2004). Repeated fires may severely diminish native annual and perennial species richness (Steers 2008). Therefore, fire and invasive species impacts on creosote bush scrub are substantial since burned stands will result in relatively poor habitat quality (Brooks & Esque 2002) and major changes to ecosystem processes (Brooks et al. 2004).

Obviously, the control and suppression of invasive annual grasses is of great importance in the conservation of creosote bush scrub. Owing to the large-scale invasion of North American deserts by invasive annual grasses, eradication seems impossible. Nevertheless, control of these invasive plants where important biological resources occur and/or in areas that are fire prone would be beneficial (see Brooks & Esque 2002; Brooks & Matchett 2006). Because fire is detrimental to ecosystem functioning, fire prevention through the control of invasive grasses would contribute the most toward creosote bush scrub preservation. If, however, that is not possible and these shrublands do burn, post-fire restoration treatments are required. Historically, fire in this arid vegetation type has been rare due to low productivity and lack of continuous fuels

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(Brooks & Minnich 2006). Consequently, limited information exists regarding restoration methods for burned creosote bush scrub.

The negative impact of fire on desert shrubs is well established (reviewed in Brooks & Minnich 2006). The impact of fire on native annual forbs is less clear. Forbs account for a large portion of the plant biodiversity in creosote bush scrub (Jennings 2001). The dearth of information may partly be due to the fact that there are hundreds of annual species associated with creosote bush scrub (Hickman 1996) and sensitivity to fire may be species specific. However, although a few native species may respond positively to fire, overall native annual plant species richness often decreases after fire (Steers 2008). A study on invasive species control was conducted to evaluate the role of invasive species in decreasing native species richness after fires, and to evaluate the ability of the native annual community to regain dominance once invasives are removed. Several treatments were implemented to control invasive annuals and observe the resiliency, or lack thereof, of native annual plants. Since most perennial plants in these desert shrublands are severely reduced by fire and are not resilient to fire disturbance (Brooks & Minnich 2006; Steers 2008), perennial plant abundance was also measured to evaluate if invasive plant removal can improve native shrub reestablishment.

Methods

Study Site

The study site (34°02′55.23″N, 116°33′57.0″W) was located in burned creosote bush scrub vegetation within Big Morongo Canyon Preserve (BMCP), which is located on the western edge of the Little San Bernardino Mountains in Morongo Valley, San Bernardino County, California, U.S.A. The study plots were placed in flat to moderately sloped (0–15°) terrain at an elevation of about 780 m. BMCP is located near the western-most extent of creosote bush scrub in the Little San Bernardino Mountains, and it is also in a transition zone between the Colorado Desert to the South and the Mojave Desert to the North.

The fire at the study site occurred on 22 June 2005. This accidental, human caused fire was named the "Paradise Fire" and it burned about 24 km^2 in Morongo Valley and adjacent hillsides. On the basis of the shrub skeletons and/or resprouts in the burned stand, and plants in an adjacent small patch of unburned vegetation opposite a fuel break, the pre-fire plant community was creosote bush scrub (see Holland & Keil 1995) co-dominated by creosote bush (*Larrea tridentata*) and white bursage (*Ambrosia dumosa*). Other shrubs included catclaw (*Acacia greggii*), Acton's encelia (*Encelia actoni*), beavertail cactus (*Opuntia basilaris*), silver cholla (*Opuntia echinocarpa*), indigo bush (*Psorothamnus arborescens*), Parish's viguiera (*Viguiera parishii*), and Mojave yucca (*Yucca schidigera*), among others. The annual vegetation in the adjacent unburned stand was dominated by the invasive grass, *Bromus madritensis* ssp. *rubens* (red brome) in interspaces and especially in shrub understories, which likely was the primary fuel in the wildfire. Other invasive annual grasses, specifically *Schismus* spp., represented mostly by *Schismus barbatus* and then *Schismus arabicus* (collectively called Mediterranean split grass), were also abundant in the interspaces. The invasive annual forb, *Erodium cicutarium* (redstem filaree), was also common. Species nomenclature follows Hickman (1996).

Average precipitation in the adjacent town of Morongo Valley is 19.6 cm with an average snowfall of 8.4 cm (WRCC 2008). Approximately 85% of the average precipitation occurs from October through April during the winter, wet season. In October 2005, a new weather station was established at BMCP where weather data during the study period were obtained. Precipitation during the experiment was 15.6, 1.2, and 18.9 cm from October through April during the first three wet seasons (2005–2006, 2006–2007, and 2007–2008). Total precipitation in the dry seasons (May through September) following the fire were very low, with scarce amounts in 2005 (D. Zeller 2005, Big Morongo Canyon Preserve, California, U.S.A., personal communication), and only 0.25 and 0.14 cm of rainfall in 2006 and 2007, respectively.

Precipitation in the first wet season was below average and irregularly distributed, resulting in mid-season drought periods (Fig. 1) that stressed the annual plants, causing some to die prematurely (R. Steers 2006, personal observation). In the second season, precipitation was insufficient for the germination of annual plants, and no invasive or native annuals were recorded (Fig. 1). Precipitation in the third and final season (2007–2008) was slightly below average with no significant droughts between precipitation events (Fig. 1). Consequently, annual plants appeared non-stressed from germination to flowering in the last season of this study.

Experimental Design

Three treatments were utilized: herbicide (H), herbicide plus weeding of invasive forbs (W), and a raking treatment (R). The three treatments and a control (C) were implemented in a randomized, complete block design composed of 12 blocks. Four plots within each block were randomly assigned to treatments H, W, R, or C. Plots were 8 × 8 m and centered on a mature, burned *L. tridentata* individual. Plots were designed to allow for the sampling of burned interspace and understory microhabitats. Two 0.5 m^2 (1.0 × 0.5 m) permanent sampling quadrats were marked with wooden stakes in interspace habitat at the two corners of the plots least influenced by shrubs, fertile islands, or disturbances. Another two 0.5 m^2 permanent quadrats were demarcated in understory microhabitat, under burned branches of the plot-central *L. tridentata* on the North and South side of the shrub. The two permanent quadrats per interspace and understory microhabitats were used to collect vegetative data during the 3-year study, whereas treatments were applied to the large plots. We applied treatments once a year to these plots during the first three post-fire growing seasons but no annual plants germinated in the second post-fire season so treatments were not applied that year (2006–2007).

Control (C) plots were left unmanipulated. The entire 8 × 8 m area of the H and W plots was broadcast sprayed with the

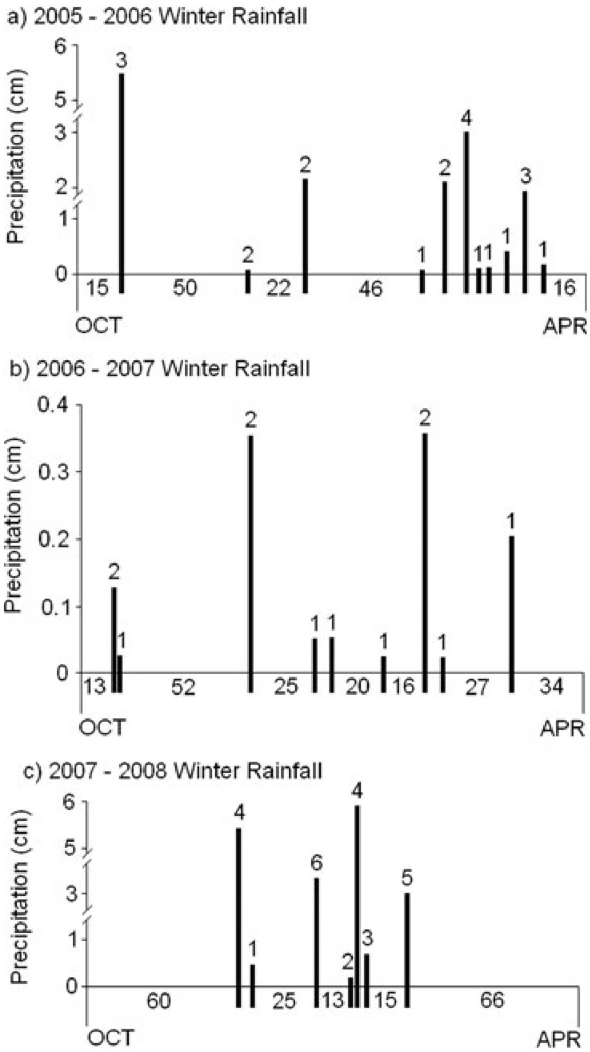

Figure 1. Winter, wet season precipitation events for 2005–2006 (a), 2006–2007 (b), and 2007–2008 (c). Numbers above bars indicate how many consecutive days of precipitation occurred while numbers below x-axis and in between bars indicate the number of consecutive days without rain that were greater than 10. Note difference in precipitation scale between years. X-axis starts on 1 October and ends on 30 April.

post-emergent, grass-specific herbicide, Fusilade II (Syngenta, Greensboro, North Carolina, U.S.A.), at a rate of 15 ml/64 m^2. Herbicide Helper (Monterey Lawn and Garden Products, Inc, Fresno, CA, U.S.A.) was used as a surfactant, also at the rate of 15 ml/64 m^2. H and W plots were sprayed with Fusilade II on 13 January 2006 and again on 7 January 2008 over the same exact areas that were treated in 2006. In W plots, *E. cicutarium*, the only non-native forb encountered, was also weeded to complement the grass-specific herbicide in removing all invasive annuals. Most of the tap root was usually removed during the weeding. Weeding in 2006 and 2008 was performed about 1 week after herbicide spraying, coinciding with the estimated time of grass mortality based on label specifications. Rather than weeding the entire 8 × 8 m plots, invasive forbs (*E. cicutarium*) were weeded by hand from W plots in two 1.5 × 1.5 m^2 that each encompassed the two 1 × 0.5 m (0.5 m^2) permanent quadrat locations in the interspace habitat. In the understory, weeding was done in a 1.5 × 0.75 m^2 that encompassed the 0.5 m^2 permanent quadrat locations under the North and South sides of the central, burned *L. tridentata* canopy.

The raking (R) treatment was implemented about 2 weeks after the first major rain event of each season, on 3 November 2005 and 14 December 2007, and it involved raking out all germinated annuals in the upper soil layer regardless of their identity. Previous observations of flowering phenology at other sites (Burk 1982; Jennings 2001) combined with observations at this site indicated that exotic species typically germinate earlier than natives, suggesting that carefully timed raking would selectively affect exotics. A scuffle hoe was used to rake the plants, and the soil was disturbed to 5 cm depth. The entire area within and immediately surrounding a permanent quadrat was raked with the scuffle hoe, regardless of whether there were visible germinated plants present, creating an additional 0.5 m buffer of R treatment around every interspace and understory quadrat.

Vegetation at Treatment Application Time

Interspace baseline vegetative parameters were collected at treatment application time each year a treatment was applied. In H, W, and R plots, percent cover of invasive grass, invasive forb, and native annuals was measured when treatments were implemented. Cover in control plots was also recorded in 2008 at the time herbicide was applied. These measures were taken with 0.5 m^2 gridded sampling frames that were placed in one of the two permanent interspace sampling frame locations per plot. In addition, the phenological stage of *Schismus* spp. and *E. cicutarium* was also recorded for the H treatment. This was done with twelve 0.125 m^2 (25 × 50 cm) frames placed in the middle of the larger 0.5 m^2 frames used to collect cover data when plots were sprayed. Phenology was recorded as the percent of *Schismus* spp. or *E. cicutarium* individuals that were in fruiting and/or flowering stage.

Peak Season Vegetative Sampling

All peak season annual plant measures were taken in late March to early April during 2006 and 2008. Absolute percent cover by species and plant species richness were measured in the 0.5 m^2 quadrats (gridded sampling frames). Cover was visually estimated per species to the nearest percent if values were 3% or greater and to the nearest 0.1% if values were below 3%. Plant density was subsampled in 0.125 m^2 (25 × 50 cm) frames within the 0.5 m^2 quadrats.

Biomass was assessed by clipping vegetation in 12 additional 0.125 m^2 frames per treatment and microhabitat in areas surrounding quadrat locations that were treated identically. Areas where biomass sampling took place were marked to avoid future sampling there. For all biomass clippings, the percent cover of species was also recorded in the 0.125 m^2 frame. A regression relationship between biomass (y-axis)

Table 1. Average percent cover of annual species found in treatment plots (C = control, R = raked, H = herbicide [Fusilade II], and W = herbicide plus weeded) in 2008.

		Interspace				Understory			
Family	*Species*	*C*	*R*	*H*	*W*	*C*	*R*	*H*	*W*
Invasive annual forb									
Geraniaceae	*Erodium cicutarium*	11.2	2.9	0.3	0.2	7.9	1.6	0.9	0.1
Invasive annual grasses									
Poaceae	*Bromus madritensis*	0.5	2	<0.1		3.7	6.6	0.3	<0.1
	Bromus tectorum		<0.1						
	Schismus spp.	39.6	40.9	0.1	<0.1	56.4	58.5	0.2	0.2
Native annuals[a]									
Asteraceae	*Calycoseris parryi*	0.7	<0.1	2.9	2.5	1.4	1.1	3.4	2.2
	Chaenactis fremontii	16.8	2.2	38.5	33.3	26.5	9.1	39.8	57.4
	Coreopsis californica		<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	0.3
	Filago depressa			<0.1	<0.1				0.1
	Malacothrix glabrata	0.5		<0.1	0.6	0.3	0.6	<0.1	2.3
	*Rafinesquia neomexicana**				0.4				
	*Stephanomeria exigua**	<0.1			0.5				
Boraginaceae	*Amsinckia tessellata**					0.3			
	Cryptantha barbigera	0.4	<0.1	3.9	3.5	3.2	2.9	10.5	7.9
	Pectocarya heterocarpa	0.1	<0.1	0.8	0.9				
	Pectocarya linearis	0.9	0.1	1.2	7.3	0.8	0.2	3.5	6.4
	Pectocarya penicillata	<0.1	<0.1	<0.1	1.1	0.2	<0.1	1.2	3.2
	Pectocarya recurvata	<0.1		0.7	1.2	<0.1		0.3	1.5
Brassicaceae	*Descurainia pinnata**			<0.1			<0.1	<0.1	0.2
	Guillenia lasiophylla	<0.1	<0.1		0.2	0.2	<0.1	0.4	
	Lepidium lasiocarpum	0.1	0.2	0.5	0.4	0.3	0.1	0.2	0.5
Fabaceae	*Lotus strigosus*	<0.1	<0.1	0.2	0.1	<0.1	<0.1	<0.1	0.2
	Lupinus bicolor	0.4	<0.1	0.6	<0.1	<0.1	<0.1	<0.1	<0.1
	Lupinus sparsiflorus					<0.1	0.1	<0.1	<0.1
Hydrophyllaceae	*Emmenanthe penduliflora*					<0.1	0.5	0.8	<0.1
	Phacelia distans		0.2	<0.1	<0.1	1.1	0.8	8.1	0.3
Lamiaceae	*Salvia columbariae*	0.3	0.3	3.4	1.5	1.1	0.5	6.1	1
Loasaceae	*Mentzelia sp.*		<0.1	<0.1			<0.1	0.7	0.2
Onagraceae	*Camissonia bistorta*		<0.1	0.7	<0.1				
	Camissonia californica	0.4	<0.1	0.9	0.6	0.4	1.2	4	1
	Camissonia pallida	<0.1	<0.1	0.4	0.8		0.2	0.2	0.5
Poaceae	*Vulpia octoflora*	<0.1	<0.1	<0.1	0.1	<0.1	<0.1	0.2	0.8
Polemoniaceae	*Eriastrum diffusum*	<0.1	<0.1	0.1	0.2		<0.1	<0.1	<0.1
Polygonaceae	*Chorizanthe brevicornu*	1.7	<0.1	1.5	0.4	<0.1		0.4	0.2
	Eriogonum maculatum				0.7				<0.1
Portulacaceae	*Calyptridium monandrum*	0.2	<0.1	0.8	1.7	<0.1	<0.1	<0.1	0.3

[a] Native species with less than 0.1% cover in all treatments are not shown, such as *Centrostegia thurberi*, *Chorizanthe watsonii*, *Crassula connata*, *Cryptantha angustifolia**, *Cryptantha maritima**, *Filago californica*, *Gilia stellata**, *Linanthus bigelovii**, *Loeseliastrum schottii**, *Lupinus concinnus*†, *Plantago patagonica*, *Stephanomeria virgata*†, *Stylocline gnaphaloides*†, *Thysanocarpus laciniatus*, *Tropidocarpum gracile**, and *Uropappus lindleyi**. Daggered (†) and asterisked (*) species were only found in 2006 or 2008, respectively.

and cover (x-axis) was then developed for the most dominant invasive grass (*Schismus* spp.), invasive forb (*E. cicutarium*), and native forb (*Chaenactis fremontii*) for each treatment × microhabitat × year.

For all cover to biomass calculations, the slope equation ($y = mx + b$) was used and the y-intercept (b) was set to zero. The slope (m) was determined using linear regression of cover to biomass from data collected in the 0.125 m^2 frames. All regressions were significant ($p < 0.05$). The following slope values were taken from regression lines and used to calculate species biomass based on species coverage in the larger (0.5 m^2) quadrats: For *Schismus* spp. in 2006, $m = 0.242$, 0.242, 0.058, and 0.242 for interspace treatments H, W, R, and C, respectively, and $m = 0.271$, 0.271, 0.058, and 0.293 in 2008. Understory *Schismus* spp. had $m = 0.269$, 0.269, 0.481, and 0.269 in 2006, and $m = 0.190$, 0.190, 0.350, and 0.236 in 2008, for treatments H, W, R, and C, respectively. Slope values (m) are reported in the same order, by treatments H, W, R, and C, for all remaining species and microhabitats. For *E. cicutarium* in 2006, $m = 0.332$, 0.332, 0.313, and 0.420 for interspace, and $m = 0.251$, 0.251, 0.248, and 0.247 in 2008. Understory *E. cicutarium* was not collected. For *C. fremontii* in 2006, $m = 0.401$, 0.373, 0.701, and 0.430 for interspace, and $m = 0.444$, 0.444, 0.276, and 0.444 in 2008. Understory *C. fremontii* had $m = 0.492$, 0.601, 0.705, and 0.620 in 2006, and $m = 0.357$, 0.299, 0.225, and 0.339 in 2008.

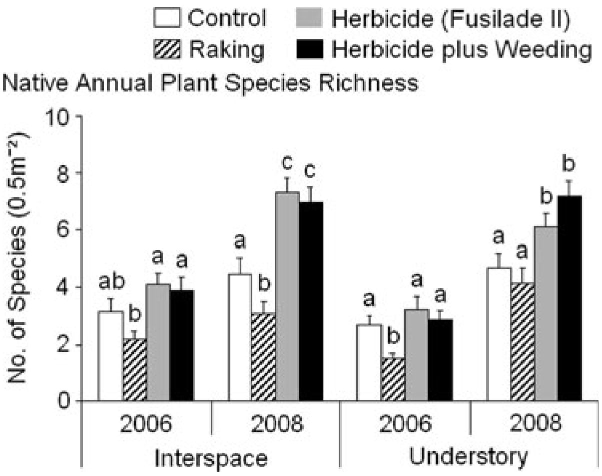

Figure 2. Average native annual species richness at peak spring season by treatment, year, and microhabitat. Different letters between bars indicate significant differences between treatments within the same year and microhabitat only, based on ANOVA and LSD tests at $\alpha = 0.05$.

Data Analysis

One-way analysis of variance (ANOVA) with a least significant difference (LSD) test for multiple comparisons was used to compare cover, richness, density, and biomass values among the treatments or between microhabitats (interspace vs. understory). Data were transformed using arcsine ($\sqrt{x}$) for cover, $\sqrt{(x + 0.5)}$ for density and species richness, and $\log_{10}(x + 1)$ for biomass when they would improve normality based on tests for Goodness-of-Fit. Kruskal–Wallis tests were used to compare perennial plant cover between treatments since these data were severely nonnormal even after transformation. JMP 7.0.2 (SAS Institute, Inc., Cary, NC, U.S.A.) software was used to conduct all of these analyses at $\alpha = 0.05$.

Results

During the three growing seasons, 4 invasive annual and 46 native annual species were recorded (Table 1). In 2008, there was greater interspace and understory native species richness in terms of average number of species per unit area (Fig. 2) and total number of unique species recorded (Table 1). Native species richness in control plots did not differ between interspace and understory microhabitats for both 2006 and 2008 based on Student *t* tests (data not shown). Also, some species documented in 2006 were absent in 2008, and vice versa (Table 1). Perennial herbs and woody shrubs totaled 18 species, but most had very low frequency and cover (Table 2). The most common perennial encountered was *Lotus scoparius* var. *brevialatus*.

Vegetation at Treatment Application Time

When the R treatment was implemented in the first post-fire season on 3 November 2005, invasive grasses were abundant at 18.8% cover, and invasive forbs and native annuals had very low absolute cover, at 0.3 and 0.2%, respectively (data not shown). By 13 January 2006, when H and W plots were sprayed with Fusilade II, invasive grass cover had increased greatly, reaching 48.7 or 74% of what the maximum cover would be in control plots measured at peak season. By this time, invasive and native annuals had also increased. Invasive forb cover was 3.4 or 74% of peak season cover, whereas native annual cover was 4.7%, only 37% of what it would reach at peak season in control plots. At the time of herbicide spraying, 61% of *Schismus* spp. and 35% of *Erodium cicutarium* individuals were already in flowering stage.

When R plots were raked again on 14 December 2007 in the third and final season, invasive grasses were less abundant than in 2006, at 9.1% absolute cover, whereas invasive forb and native annual cover was greater than in 2006, at 2 and 0.8%, respectively. When H and W plots were sprayed with herbicide in 7 January 2008, cover of invasive grass, invasive forb, and native annuals in control plots was only 5.5, 3.8, and 2.4%, respectively, or 14, 34, and 11%, respectively, of what maximum peak season cover would be later in the year for control plots. This revealed that these plots were treated at a much earlier growth stage than in 2006. Just as importantly,

Table 2. Average native perennial cover in 2008 for all treatments (C = control, R = raked, H = herbicide, and W = herbicide plus weeded).

		Interspace				*Understory*			
Family	*Species*[a]	*C*	*R*	*H*	*W*	*C*	*R*	*H*	*W*
Asteraceae	*Encelia actoni*					0.2			0.2
	Ambrosia dumosa		0.8	<0.1					
	Viguiera parishii					0.5			
Fabaceae	*Acacia greggii*		0.2					<0.1	
	*Psorothamnus arborescens**		0.5						
	Lotus scoparius	<0.1	0.3	1	0.3	0	<0.1	<0.1	<0.1
Krameriaceae	*Krameria grayi*					<0.1	<0.1	0.1	
Liliaceae	*Dichelostemma capitatum*	<0.1	<0.1	0.3	0.3	<0.1		<0.1	<0.1
Malvaceae	*Sphaeralcea ambigua*				0.2		1.8	0.2	
Nyctaginaceae	*Mirabilis bigelovii*		<0.1	<0.1		2.9	<0.1	1.6	1.5
Poaceae	*Pleuraphis rigida*		<0.1			0.1			

[a] Species with less than 0.1% in all treatments are not shown, including *Chamaesyce albomarginata*†, *Larrea tridentata**, *Salazaria mexicana**, *Selaginella bigelovii*†, *Stephanomeria pauciflora*, *Stillingia linearifolia*, and *Yucca schidigera*†. Daggered (†) and asterisked (*) species were only found alive in 2006 or 2008, respectively.

no *Schismus* spp. or *E. cicutarium* individuals had reached fruiting and/or flower stage at herbicide application time in 2008.

Treatment Effects on Annual Vegetation

Interspace. In 2006, treatments R, H, and W reduced invasive annual grass cover by about half (Fig. 3). In 2008, H and W treatments reduced invasive grass cover to almost 0% while the raking treatment had no effect. In 2006, treatment W reduced invasive forb cover, but in 2008, when precipitation was more favorable and Fusilade II was sprayed at an earlier phenological stage, both H and W treatments were equally effective at reducing invasive forb (*E. cicutarium*) cover. Although all treatments were effective at reducing invasive grass in 2006, native annuals had no positive response in this year and did not differ from the control. However, in 2008, the H and W treatments had greater native annual cover than control and raking treatments. In fact, the raking treatment actually reduced native annual cover in 2008 (Fig. 3). Native annual plant species richness responded similarly to native cover measures, although reduced native richness in the R treatment was already evident by 2006 (Fig. 2).

All treatments reduced invasive grass density in 2006 (Fig. 4), with the lowest densities occurring in the R treatment. By 2008, invasive grass density in the R treatment was equal to the control while the densities in H and W plots were very low, implying excellent herbicide effectiveness. Invasive annual forb density was reduced by all three treatments, especially for H and W plots in 2006 and 2008 (Fig. 4). Raking severely reduced native annual density in both years. However, due to high variance in 2008, there was no decrease compared with control (Fig. 4). In 2008, treatments H and W increased native annual density.

Invasive grass (*Schismus* spp.) biomass was reduced by treatments H and W in 2006, and then by all three treatments in 2008 (Table 3). Invasive forb (*E. cicutarium*) biomass had a similar response as invasive grass biomass except only treatment W was effective in 2006. Native forb (*Chaenactis fremontii*) biomass increased in the R treatment in 2006, but had an opposite response in 2008. Only the H and W treatments in 2008 increased native forb biomass (Table 3).

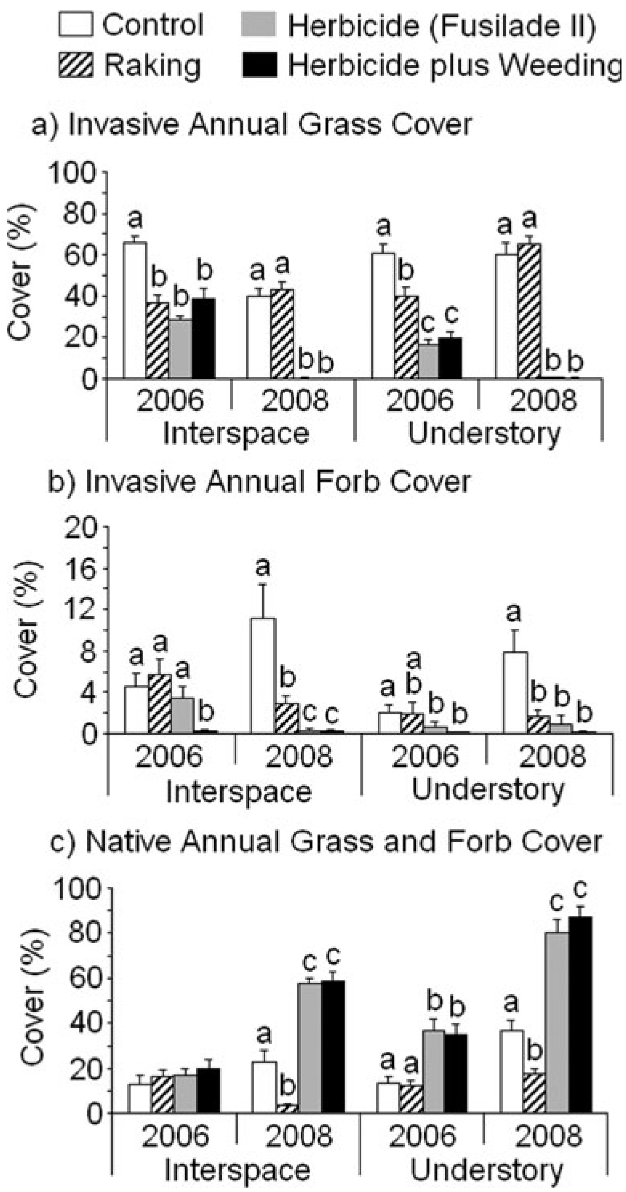

Figure 3. Average cover of invasive grass (a), invasive forb (b), and native annuals (c) at peak spring season. Different letters between bars indicate significant differences between treatments within the same year and microhabitat only, based on ANOVA and LSD tests at $\alpha = 0.05$.

Understory. In 2006, treatments had a stronger effect on understory annual plant cover compared with the interspace (Fig. 3). Treatments induced positive responses in native annual cover compared with control plots in 2006, in contrast to the interspace microhabitat. The same pattern was observed in 2008. Differences in understory native annual richness between treatments were similar to those observed for understory native annual plant cover, both in 2006 and 2008 (Figs. 2 & 3). Treatments utilizing Fusilade II clearly were best at reducing invasive grass and forb cover while increasing native annual cover and richness (Figs. 2 & 3).

Density of invasive grasses and forbs was reduced by all treatments in 2006 and by some in 2008; the raking treatment had no effect on invasive grasses in 2008 (Fig. 4). In 2006, treatments had little effect on native annual density but in 2008, the R treatment reduced native annual density while the W treatment increased this parameter.

Invasive forb (*E. cicutarium*) biomass was not measured in the understory microhabitat since it was uncommon in this microhabitat in 2006 when biomass sampling was first initiated (Figs. 3 & 4). However, invasive grass (*Schismus* spp.) and native forb (*C. fremontii*) biomass were sampled in the understory (Table 3). Treatments W and H were equally effective at reducing invasive grass biomass in 2006 and 2008, but only treatment W increased native forb biomass in

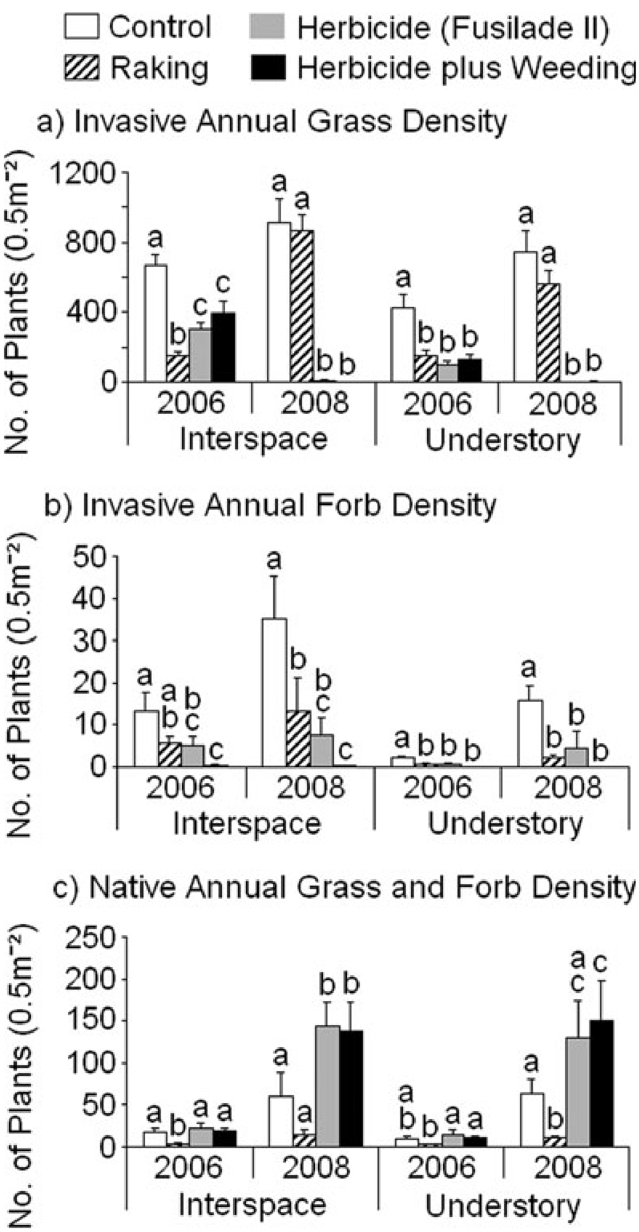

Figure 4. Average density per 0.5 m^2 of invasive grass (a), invasive forb (b), and native annuals (c) at peak spring season. Different letters between bars indicate significant differences between treatments within the same year and microhabitat only, based on ANOVA and LSD tests at $\alpha = 0.05$.

both years. The R treatment decreased understory native forb biomass in 2008.

Treatment Effects on Perennial Vegetation

Interspace perennial plant cover in control plots decreased from $0.22 \pm 0.12\%$ in 2006 to $0.08 \pm 0.04\%$ in 2008 but increased in the understory microhabitat from $2.51 \pm 2.02\%$ in 2006 to $3.73 \pm 3.44\%$ in 2008. Perennial cover was usually greater in treated plots compared with control, but no treatment effects were found based on Kruskal–Wallis tests when all treatments were compared simultaneously. However, when comparing perennial plant cover or density between each treatment and the control separately, some treatments produced greater native perennial cover or density than the control. Treatment R had greater perennial plant cover ($1.82 \pm 0.82\%$) than the control for interspace microhabitat in 2008 ($\chi^2 = 4.8093$, $p = 0.0283$). Treatment W also had greater cover ($1.2 \pm 0.4\%$) than control for interspace microhabitat in 2008 ($\chi^2 = 5.4996$, $p = 0.0190$), and greater perennial density (3.83 ± 1.96 individuals/0.5 m^2) than control (0.17 ± 0.17 individuals/0.5 m^2) in 2006 ($\chi^2 = 5.4679$, $p = 0.0194$) and in 2008 (7.3 ± 3.1 vs. 0.83 ± 0.83 individuals/0.5 m^2) ($\chi^2 = 4.9647$, $p = 0.0259$).

Discussion

On the basis of the observations of adjacent unburned vegetation, the most obvious effect of the fire was the dramatic reduction in abundance of desert shrubs (approximately 25% pre-fire cover to <1% post-fire, live shrub cover). Although some shrub individuals were able to resprout after the fire, shrub cover remained very low throughout the experiment. Some shrubs, such as *Lotus scoparius*, appeared to increase after the fire due to seedling recruitment, an expected response based on observations from neighboring cismontane California shrublands (Keeley 1991). *Bromus madritensis* abundance was dramatically reduced by the fire, but *Schismus* spp. and *Erodium cicutarium* benefited, similar to other studies (Minnich & Dezzani 1998; Brooks 2002).

Treatment Effects on Annual Vegetation

Success of the raking treatment was highly dependent on the annual plant species group (invasive grass, invasive forb, or native annual) composition at treatment time. In 2006, when precipitation was early in the season, the majority of emerged annuals were invasive grasses, and the raking treatment disproportionately impacted this species group. In 2008, precipitation came later in the growing season and invasive grasses were not as abundant as in 2006, but invasive and native forbs were more abundant. Consequently, the R treatment had a greater negative impact on invasive and native forbs than invasive grasses in 2008 compared with 2006.

Mechanical control techniques like scuttle hoe application have produced successful results for controlling invasive plants in rangelands (reviewed in DiTomaso 2000). However, this study showed that the scuttle hoe treatment also negatively impacted native annual plants. Non-specific treatments such as raking could be useful in certain instances when invasive plants are vulnerable and desired plants are not, such as when invasive species germinate earlier than natives. However, this particular treatment also resulted in major soil disturbance, which can promote invasive plants (reviewed in Hobbs & Huenneke 1992). Use of a general herbicide, such as glyphosate, could have achieved similar plant mortality rates as

Table 3. Average biomass (g/0.5 m^2 ± SE) for *Schismus* spp., *E. cicutarium*, and *C. fremontii* in each of the four treatments (C = control, R = raked, H = herbicide, and W = herbicide plus weeded).

	Interspace			*Understory*	
	Schismus	*Erodium*	*Chaenactis*	*Schismus*	*Chaenactis*
2006					
C	61.9 ± 4^{A}	7.8 ± 2^{A}	12.7 ± 4.5^{A}	63.2 ± 4.3^{A}	17.9 ± 5.5^{A}
R	50.9 ± 5.7^{A}	7.1 ± 1.9^{A}	28.3 ± 6.6^{B}	66.1 ± 10.2^{A}	22.2 ± 6.7^{AB}
H	22.0 ± 1.7^{B}	4.5 ± 1.6^{A}	13.7 ± 2.3^{AB}	14.1 ± 2.2^{B}	34.4 ± 7.8^{AB}
W	29.1 ± 4.1^{B}	0.4 ± 0.2^{B}	14.4 ± 3.4^{AB}	16 ± 2.4^{B}	41.7 ± 8.7^{B}
2008					
C	153.9 ± 14^{A}	11.1 ± 3.2^{A}	29.8 ± 7.8^{A}	53.2 ± 6.9^{A}	35.9 ± 6.2^{A}
R	9.5 ± 1^{B}	2.9 ± 0.9^{B}	2.5 ± 1^{B}	81.9 ± 7.2^{B}	8.2 ± 1.4^{B}
H	0.1 ± 0.1^{C}	0.3 ± 0.2^{C}	68.5 ± 6.8^{C}	0.1 ± 0.1^{C}	56.8 ± 9.8^{AC}
W	0 ± 0^{C}	0.2 ± 0.1^{C}	59.2 ± 7.8^{C}	0.1 ± 0.1^{C}	77.8 ± 8.4^{C}

Different letters indicate differences between treatments within year, species, and microhabitat only, based on ANOVA and LSD tests at $\alpha = 0.05$.

the scuttle hoe application but with no soil disturbance, which would have better tested this phenology-based approach (see Brooks et al. 2006).

Fusilade II was a very effective product for controlling invasive annual grasses and had the surprising additional effect of controlling *E. cicutarium*. This product has successfully reduced invasive grass abundance in many instances (Pavlick et al. 1993; Arnold et al. 1998; Cione et al. 2002; Allen et al. 2005; Cox & Allen 2008; Marushia & Allen 2009). We observed that mid-season droughts and consequent plant stress can have a great effect on the success of Fusilade II. In 2006, precipitation was below average and irregular in distribution; grass and *Erodium* mortality in response to herbicide was low. In 2008, precipitation was greater and rainfall events occurred closer together, precluding a period early in the growing season where drought stress could have affected herbicide response by the invasive grasses and forbs. Also, timing of herbicide application was critical for inducing mortality of invasive grasses and *E. cicutarium*. In 2006, 61% of *Schismus* spp. and 35% of *E. cicutarium* were already flowering when plots were sprayed while in 2008 plants were at an earlier growth stage overall and no invasive plants were flowering yet. Consequently, in 2008, Fusilade II was as effective as hand pulling for removing *E. cicutarium*.

Fusilade II was able to reduce the invasive forb, *E. cicutarium*, in addition to invasive grasses. This effect is very promising for the restoration of shrublands because this invasive forb is ubiquitous and abundant in the deserts of North America (Hickman 1996; Brooks & Esque 2002; Schutzenhofer & Valone 2006), and in other arid to semiarid environments including cismontane California, Chile, and Australia (Westbrooke et al. 1998; Figueroa et al. 2004; Cox & Allen 2008). Although the ability of Fusilade II to kill *Erodium* spp. is not indicated on the product label, the mechanism by which this occurs may be similar to that of another closely related herbicide, Haloxyfop, which has been shown to kill both grasses and a couple of genera in the Geraniaceae, including *Erodium* (Christopher & Holtum 2000).

No evidence of negative effects on native species from Fusilade II was observed although extreme caution must be used when applying this product, especially where native grasses or native *Erodium* species occur. *Vulpia octoflora*, a native annual grass that did occur occasionally in plots treated with Fusilade II, did not appear to be damaged. This may be due to Fusilade II herbicide resistance by this genus (*Vulpia*/*Festuca* sensu lato) (Catanzaro et al. 1993; Yu et al. 2004) or due to phenological differences in germination or flowering time that allowed *V. octoflora* to avoid vulnerability during application.

Invasive annuals were shown to suppress the cover, density, biomass, and species richness of native annual plants in this burned shrubland. Once invasive species were removed, native annuals responded positively in all of these measures. Fire is known to increase invasive annual plants and reduce the abundance and richness of native annuals in desert shrublands close to this study site (Steers 2008). The results from this study suggest that such reductions are largely due to displacement of natives by invasive annuals, and not due to major loss of the native annual seed bank from the fire, as native forb density increased following control of invasives. However, high fire temperatures in shrub understories may limit viable propagules of native plants in that microhabitat (Brooks 2002), so fire impacts on seed banks cannot be completely discounted.

Invasive plants also occur extensively in unburned desert shrublands, and similar positive responses by native forbs to invasive plant control have been documented there (Steers 2008). Invasive plant control in unburned stands would also greatly reduce or eliminate the possibility of future fire disturbance (Brooks et al. 2004), which devastates the shrub components and further promotes invasive annuals (Brooks 2002; Steers 2008). The conservation of desert shrublands in fire-prone regions should focus on invasive plant control in both remaining unburned stands and burned stands, realizing that the greatest positive impact toward shrubland conservation will come from a fire prevention strategy based on invasive plant control.

Treatment Effects on Perennial Vegetation

Statistically greater cover and density of perennial plants in interspace weeded (W) plots compared with the control lends evidence for the competitive suppression of desert shrub seedlings by invasive annuals. Although invasive grass competition with mature shrubs has been documented in creosote bush scrub (DeFalco et al. 2007), we are not aware of other studies in this vegetation type that document competition with invasives at the shrub seedling stage. However, shrub and tree seedlings from other semiarid vegetation types are competitively suppressed by similar invasive annual grasses (Schultz et al. 1955; Davis & Mooney 1985; Gordon et al. 1989; Eliason & Allen 1997; Pendelton et al. 2007). Use of Fusilade has also been successful in establishing shrub seedlings in coastal sage scrub impacted by fire and invasive annual grasses (Cione et al. 2002).

For burned understory microhabitat, perennial plant abundance was very low and treatments had no effect on perennial plant cover or density, possibly due to propagule limitation. Relatively high fire temperatures in the understory compared with the interspace may have limited viable perennial seeds in this microhabitat (sensu Brooks 2002). Many desert shrubs are not adapted to fire (Brooks & Minnich 2006), and their seeds may also be highly sensitive to fire. This may explain why invasive plant removal from understories did not result in increased perennial plant density or cover despite the pre-fire potential for high shrub seed density in the understory (Nelson & Chew 1977). Alternatively, the duration of this study may have not been long enough to adequately document the perennial plant response to invasive plant control in the understory.

Conclusion

Most restoration efforts in desert shrublands of North America have focused on perennial species and cases of severe physical disturbances (e.g., roads, mining, and farming) (Grantz et al. 1998; Lovich & Bainbridge 1999; Walker & Powell 2001; Bainbridge 2007) or on riparian ecosystems that have been invaded by *Tamarix* spp. (tamarisk) (Fleishman et al. 2003; Shafroth & Briggs 2008). Examples of restoration of creosote bush scrub-annual communities from exotic-annual invaded states are lacking. Native annuals are important components of species diversity (Jennings 2001) and ecosystem processes (Brooks et al. 2004). They are also important for eco-tourism because they produce colorful, episodic flower displays (Cowling 1999; Weaver 2001). Removal of invasive grasses and forbs is a logical first step in desert restoration since they displace native annual and perennial species, and promote fire.

The control of invasive annuals in exotic/native species mixtures without non-target effects is challenging. In this study, the Fusilade II treatment was most effective for reducing invasive plant abundance and promoting the native plant community. Invasive plant control also increased the establishment of perennials in post-fire vegetation, which is critical for reinstating the structural complexity of desert shrublands (Bolling & Walker 2002).

Implications for Practice

- The use of Fusilade II, a grass-specific herbicide, can effectively control invasive annual grasses and the invasive annual forb, *Erodium cicutarium*, two extremely problematic plant groups in the deserts of North America and elsewhere.
- Post-fire invasive annual plant control increases the abundance and species richness of native annual plants.
- Once invasive annuals are removed, passive perennial plant recruitment also increases, which might lessen the need for costly shrub restoration treatments.
- Treatments that are non-species specific are highly constrained due to potential non-target effects. In this study, the non-selective treatment, raking, performed poorly compared with the species-specific treatments, herbicide and herbicide plus weeding, for promoting native vegetation.
- Limiting invasive grass abundance should consequently reduce the likelihood of future fire (Brooks et al. 2004), decoupling the invasive plant–fire feedback that leads to vegetation type conversion.

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