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RESEARCH ARTICLE

Simple Approaches to Improve Restoration of Coastal Sage Scrub Habitat in Southern California

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ABSTRACT: Much of the coastal sage scrub habitat in Southern California that existed prior to European settlement has been developed for human uses. Over the past two to three decades, public agencies and land conservation organizations have worked to acquire some of the remaining lands for preservation. Many of these lands are degraded by past intensive livestock grazing, farming, and frequent fires, and the native flora has been replaced by weedy, exotic annual grasses and forbs, mostly of Mediterranean origin. Restoration of native flora is challenging and there are few successful examples to provide guidance on effective methods. Cost is also an important and prohibitive factor. Competition from weeds is one of the most difficult impediments to establishing native vegetation, which often persists in the seedbank. We compared annual applications of the nonselective herbicide glyphosate over multiple years, followed by a final year with the grass-specific fluazifop, as a simple, low cost method of reducing the exotic seedbank sufficiently to allow native vegetation to establish. This approach was combined with seeding native forbs, herbaceous perennials, and shrubs in one half of each treatment plot. Herbicide treatments were made in the spring each year from 2006 to 2010, and were combined with weed trimming in 2010 and 2011 to remove exotic forb inflorescences, and raking to remove litter. In 2010, native plant cover in herbicide-treated plots was about 50%, consisting of 43 species, compared to <5% cover in the control plots. Most of the native plants came from the existing seedbank, and very few from the seed mix. A cost analysis showed that a once-yearly herbicide treatment was as effective as one application plus spot spraying or hand inflorescence trimming, and is more cost-effective than hand weed control and raking for restoration.

Index terms: coastal sage scrub, fluazifop, glyphosate, herbicide, weeds

INTRODUCTION

Coastal sage scrub (CSS) is a Mediterranean-type floristic community in lower elevation California and northern Baja California, Mexico (Rundel 2007). In southern California, it occurs in a coastal band from Santa Barbara on the north to San Diego in the south and inland to the Riverside area. CSS vegetation is diverse and species rich, but typically dominated by a few shrub species, such as *Artemisia californica* Less., *Eriogonum fasciculatum* Benth., *Salvia mellifera* Greene, *S. leucophylla* Greene, and *Rhus integrifolia* (Nutt.) Rothr. (Weaver and Clements 1938; Rundel 2007). Rainfall is limiting throughout this range, averaging 450 mm in the northern extent to 250 mm in the south and inland regions. Rain falls sporadically and unpredictably from fall through spring, followed by a summer drought of five or more months (Dallman 1998). Several of the dominant species are drought deciduous, including *A. californica* and the *Salvia* species. Other dominant species exhibit sclerophyllous leaves, small leaves, are succulents, or have extensive root systems.

European settlement of California resulted in extensive loss of CSS and conversion to exotic annual grassland (Minnich and Dezzani 1998; Allen et al. 2000; Talluto and Suding 2008; Fleming et al. 2009). Early Spanish colonists in the late 18th century introduced extensive livestock grazing, and management practices through the early 20th century often included intentional fire to burn off scrub and encourage forage (Burcham 1957). Most lands within the range of CSS that were level enough to till were converted to agriculture or pasture before the first statewide vegetation survey of the 1930s (Vegetation Type Map; Cox et al. 2014). Some of these lands have more recently been converted to housing developments (Chen et al. 2010), as CSS includes the most desirable coastal lands for urban expansion. The result is the significant loss of CSS vegetation. Estimates of presettlement CSS that has been converted to agriculture, urban development, and exotic annual grassland vary from 40 to 90% (Westman 1981; Cleland et al. 2016). The quality of remaining CSS communities varies considerably; estimates suggest that 50% of the remaining habitat is degraded by frequent wildfires, invasive plants, air pollution, and other causes (Freudenberger et al. 1987; Allen et al. 2000; Cleland et al. 2016).

CSS has one of the highest concentrations of federally listed and rare species in the United States, and is considered one of the most threatened ecosystems in the country (Noss et al. 1995). In an effort to preserve CSS, government agencies and private conservancies have been acquiring land as part of habitat conservation plans to prevent urban development (Scott et al. 2006). Some of this CSS habitat is undergoing restoration because the native vegetation has been replaced by exotic annuals. Former cattle ranches are typically the largest contiguous properties that have conservation value, but also include degraded lands. For example, the County of San Diego Barnett Ranch Reserve near the city of Ramona, California, where we conducted our research, was used by Mission San Diego for cattle grazing in the mid-eighteenth century. The County bought the property in 2002 to become part of the County Multi-Species Conservation Plan, a habitat conservation plan designed to satisfy mitigation concerns regarding development and endangered species (Helix Environmental Planning 2004). The entire 294-ha reserve burned in a wildfire in November 2003. Although the property was purchased for its native habitat value, approximately 95 ha is dominated by weedy exotic grasses and forbs (Helix Environmental Planning 2004). The presettlement habitat of this land is unknown, but based on fragments of remnant vegetation it is likely that it had been a mix of CSS, native perennial grassland, and native annual forb fields (Helix Environmental Planning 2004).

These exotic grasslands do not undergo natural succession to CSS, native grassland, or annual forbland (Stylinski and Allen 1999; Allen et al. 2005), and must be restored. Removal of the exotic grasses with herbicides or other methods has been used to restore CSS and native grassland (Eliason and Allen 1997; Cione et al. 2002; Gillespie and Allen 2004; Marushia and Allen 2011; Bell et al. 2013; Kimball et al. 2014), but exotic grass removal has often been transitory or has resulted in the replacement of the exotic grass with exotic forbs (Allen et al. 2005; Talluto et al. 2006). One problem might be the duration of the weed removal efforts, which was only 2–3 years in these studies. In addition to removing the extant exotic vegetation, we hypothesized that successful restoration also requires seed bank control. The exotic annual seedbank was 12,000 per m^2 in CSS with an intact shrub cover but historically grazed, while the native seed density was only 400 seeds/m^2 (Cox and Allen 2008a). If seed inputs to the soil are stopped, the weed seed bank declines rapidly (Radosevich et al. 2007). Our restoration research was designed to determine how different levels of invasive weed control would influence establishment from the native seedbank of CSS (Allen et al. 2001). For comparison, we included a subplot treatment of seeding native forbs and shrubs to enhance the native seedbank (Allen et al. 2001). Weed control was done annually using herbicides to reduce additions to the weed seed bank while allowing depletion of the existing seed. An adaptive management approach was used to control different weed life forms as the community developed over time. To encourage establishment of the native community of forbs and shrubs, the broad-spectrum herbicide used initially was substituted by a grass-specific herbicide after three years. At this point, inflorescences of exotic forbs were trimmed manually to reduce their contribution to the seedbank. If successful, this would be a simple, relatively low cost approach that could be adopted by land managers in Southern California.

METHODS

Research plots were established on two sites approximately 100 m distant from each other within exotic grass and forb-dominated areas in the Barnett Ranch Reserve. The vegetation adjacent to the research site is a subassociation of CSS known as inland, or Riversidean, sage scrub (Westman 1981). Our preliminary vegetation survey showed ~1% cover of native shrubs and forbs in a matrix of dense exotic grasses and forbs, suggesting there is still a native seedbank that might germinate upon reduction of exotic seeds. Site 1 was located on a south-facing hill with a 30% slope and was dominated by exotic forbs. The soil at Site 1 is a Cienaba very rocky coarse sandy loam—a shallow soil 10 to 50 cm deep over bedrock with a very low water availability capacity of less than 2 cm. Site 2 was 120 m to the southeast of Site 1, on flatter ground (5 to 9% slope) with a north-facing aspect. Site 2 was dominated by exotic grasses, with soil characterized as a Fallbrook Rocky Sandy Loam—a deeper soil, 50 to 100 cm above bedrock, and a water availability capacity of 18 cm. Elevation at both sites is about 450 m. Mean precipitation is 414 mm per season (July 1 to June 30), but seasonal variation over the years of this research was 152 mm in the 2006/07 season, 371 mm in 2007/08, 285 mm in 2008/09, and 441 mm in 2009/10, accompanied by considerable in-season variation (CIMIS 2014). The site has relatively low anthropogenic nitrogen deposition (~6 kg N ha^{-1} yr^{-1}, Fenn et al. 2010).

Our research utilized a split-plot design with three treatments at each of the two sites. Herbicide was the main plot treatment, applied to plots 6 × 18 m with five replications per treatment. Treatments were; (1) herbicide applied once each spring broadcast over the whole plot (hereafter Herbicide 1X), (2) herbicide applied broadcast once in the spring followed by spot spraying weeds as needed that emerged after the initial treatment (hereafter Herbicide 2X), and (3) an unsprayed control plot (hereafter control). Broadcast herbicide applications were made in February or March each year depending upon exotic plant emergence following winter rains. Broadcast herbicide applications were made with a hose-end sprayer with a boomless nozzle (Boominator 1250) pressured by an electric diaphragm pump at 275 kPa and a spray volume of 280 L ha^{-1}. A surfactant was present in all herbicide applications to improve herbicide absorption into the plant foliage. Spot spraying in the second herbicide treatment was done two weeks after the broadcast application with a hand-pressured backpack sprayer, using a 2% solution of glyphosate in water. Broadcast applications were made over the whole plot on emerged weeds and native forbs, but care was taken to avoid application to the few native shrubs (typically one or less per plot) that had survived the wildfire. The herbicide used annually in 2006 through 2009 was glyphosate, a nonselective herbicide applied at a rate of 4.7 L (0.28 kg ae) ha^{-1}.

Subplots were either seeded with a mix of seven native CSS species or nonseeded. Subplots were assigned randomly to one half of the main plots divided on the axis, 6 × 9 m. Seeding occurred in February 2007, the second season of treatment. Before seeding, all unsprayed control plots were mowed to 2–6 cm height with string trimmers to improve seed contact with soil. The mowed duff was raked off the plots to buffer areas, taking care not to rake duff across different treatments. The herbicide treated plots had scant litter or living plants so de-thatching was not needed. Commercial seed (S&S Seeds Inc.) that had been imbibed for 24 hours were hand sown following duff removal and in anticipation of rainfall. Species sown included *Eriophyllum confertiflorum* (DC) A. Gray*, Eriogonum fasciculatum, Acmispon glaber* (Vogel) Brouillet*, A. strigosus* (Nutt.) Brouillet*, Marah macrocarpa* (Greene) Greene*, Crocanthemum aldersonii* (Greene) Janch.*, and Cryptantha intermedia* (A. Gray) Greene. All seed had been collected from naturally occurring plants in the area. Rainfall was 49 mm the two weeks following sowing and the total accumulation was 78 mm the rest of the season until 1 July 2007 (CIMIS 2014). Species nomenclature follows Baldwin et al. (2012).

Because of increasing native species cover in 2010, fluazifop-P-butyl (Fusilade DX) herbicide was broadcasted 23 March 2010 at 1.3 L ha^{-1} in the herbicide treatments. This herbicide, which generally is only toxic to Poaceae, can also injure or inhibit seed production of some *Erodium* species (Christopher and Holtum 2000; Steers and Allen 2010; Weathers 2013). Since *Erodium* was the dominant genus in the treated plots, the use of fluazifop-P-butyl offered an opportunity to reduce seed production by *Erodium* spp. while avoiding injury to abundant native flora. The glyphosate spot spray treatments were also suspended in 2010 and no herbicides were applied in 2011. To assist in continued depletion of the weed seed bank, the inflorescences of weedy species, principally *Erodium* spp. and *Hirschfeldia incana* (L.) Lagr.-Fossat, were cut off with a string trimmer. This operation commenced in March 2010 and was repeated four times through the season until late May 2010.

Data collected included plant cover and species richness in one 0.5 by 1.0-m fixed quadrat per subplot in May of 2006, 2007, and 2008. In 2009, data were not collected because of fiscal limitations. Data were again collected in 2010. For analysis, cover data for species were combined into three functional groups: native forbs, exotic forbs, and exotic grasses. Pretreatment cover and richness data were collected prior to the 2006 herbicide application. A one-way ANOVA of plots within each site showed no significant differences in cover and richness of functional groups prior to treatment application. The 2006 through 2008 main plot cover data for May of each year were analyzed with a repeated measures ANOVA for each site and each plant functional group, where the repeated measure is cover of functional groups over the three years (JMP 2012). Subplot data were combined to have one sample per main plot (i.e., there were five replications). The repeated measures analysis showed significant year (repeated measure) by treatment interactions in most cases, so data were also analyzed separately by year using one-way ANOVA to compare among the three treatments. Arcsin transformations were used prior to analysis to improve normality of the data. Cover data in tables and figures are back-transformed for presentation (Steele and Torre 1980). Because of the unknown impact of a lack of glyphosate treatment in 2010, the effect of the fluazifop treatment, the 2009 gap in data collection, and the weed inflorescence removal effect, the 2010 data were analyzed separately with treatment and replication as factors. When significant differences ($P < 0.05$) existed among treatment means within a year, they were separated using the Tukey-Kramer Highly Significant Difference Test (T-KHSD), $P = 0.05$.

In 2011, an inventory of native shrubs was taken counting all individuals of five species (*A. glaber, E. confertiflorum, E. fasciculatum, A. californica*, and *Salvia apiana* Jeps.) in each of the 6 × 9-m subplots (total of 30 subplots). Factorial ANOVA was performed on count data on these five species and the combined data of the three seeded species (*A. glaber, E. confertiflorum* and *E. fasciculatum)* in Site 1. There was low shrub emergence in Site 2, and many plots had no seeded plants emerge at all, so these data were not analyzed statistically. Site 1 data were subjected to square root transformation before analysis to improve the homogeneity of variances (Steele and Torre 1980).

Records were kept for all operations relevant to treatments. These included: the amount of herbicide used; the time required to spray treated plots, both broadcast and spot spray; and time required to trim weed inflorescences. This information was used to calculate costs of restoration treatments.

RESULTS

Pretreatment cover at both sites was dominated by a small number of mostly exotic species, largely *Erodium botrys* (Cav.) Bertol. and exotic grasses (Table 1). The native plant community comprised of five species whose combined cover was less than 1% in either site (Table 1). Post-treatment cover for each functional group was affected significantly by herbicide treatment compared to controls over the three years 2006–2008, with a significant interaction of the repeated measure by treatment for native forbs in Site 2, but not Site 1 (Table 2).

Herbicide Treatment Effects on Plant Cover by Functional Group

Exotic grass cover was significantly decreased by both the 1X and 2X herbicide treatments compared to controls in both sites, and there were significant interactions of the repeated measure (cover over three years) by treatment in both sites (Table 2). Nine exotic grass species were present, nine in Site 1 and eight in Site 2. The most common species were *Bromus diandrus* Roth*, Hordeum murinum* L. and *Avena fatua* L. While the repeated measures analysis showed an overall response by grass over the three years in both sites, univariate analyses in 2006 showed that post-treatment exotic grass cover did not differ between herbicide treatments at either site ($F = 3.777$, $P = 0.07$ at Site 1 and $F = 0.504$, $P = 0.620$ at Site 2, Figures 1a and 1d). However, the treatment was effective by the second year of application, as both herbicide treatments greatly

Table 1. Pretreatment percent cover estimates (n = 30) by experimental site. Barnett Ranch Reserve, Ramona, California, March 2006.

Site	Native Species	Exotic Forbs	Exotic Grasses
1	*Artemisia californica* (0.3%)	*Hirschfeldia incana* (0.3%)	*Avena* sp. (0.2%)
	Daucus pusillus (0.08%)	*Brassica nigra* (0.1%)	*Bromus* sp. (4.4%)
	Acmispon glaber (.02%)	*Erodium botrys* (60.1%)	*Hordeum* sp. (0.03%)
		E. ciculatarium (0.5%)	
		Logfia gallica (0.01%)	
		Hypochaeris glabra (0.1%)	
		Silene gallica (0.5%)	
	Total Cover: <1%	**Total Cover: 61.2%**	**Total Cover: 5.3%**
2	*Ambrosia confertiflora* (0.1%)	*Lysimachia arvensis* (0.1%)	*Avena* sp. (0.9%)
	Pseudognaphalium californicum (0.4%)	*Convolvulus arvensis* (0.1%)	*Bromus* sp. (31.5%)
		E. botrys (6.7%)	*Hordeum* sp. (18.1%)
		Rumex crispus (0.1%)	
	Total Cover: <1%	**Total Cover: 7.0%**	**Total Cover: 49.8%**

reduced exotic grass cover compared to the unsprayed control at both sites in 2007 (F = 8.667, P = 0.01 at Site 1 and F = 203.44, P < 0.0001 at Site 2), 2008 (F = 21.349, P = 0.001 at Site 1 and F = 18.541, P = 0.001 at Site 2), and 2010 (F = 64.237, P < 0.0001 at Site 1 and F = 80.003, P < 0.0001 at Site 2, Figures 1a and 1d). Exotic grass cover was consistently higher in Site 2 (28.43±4.7%) than Site 1 (5.67±1.3%). Exotic grass cover increased annually in both sites in the unsprayed control plots (F = 27.706, P < 0.0001 for Site 1 and F = 28.558, P < 0.0001 for Site 2).

Exotic forb cover comprised 24 species, 18 in Site 1 and 22 in Site 2. The most common species were *E. botrys, Lysimachia arvensis* (L.) U. Manns & Anderb. , *H.incana* and *Silene gallica* L. Exotic forb cover was also significantly decreased by both the herbicide treatments compared to the unsprayed control in both sites, but there were significant interactions of the repeated measure by treatment (Table 2). The univariate analyses showed no significant differences in exotic forb cover among herbicide treatments in the first year (F = 2.235, P = 0.169 Site 1 and F = 1.04, P = 0.396 in Site 2, Figures 1b and 1e). In the second and third years (2007 and 2008), herbicide treatments greatly reduced exotic forb cover compared to the control (F = 36.323, P < 0.0001 in 2007 and F = 11.839, P = 0.004 in 2008) in Site 1 (Figure 1b). In Site 2, Herbicide 2X exotic forb cover was not different from the control in 2007 (Figure 1e). Conversely, in 2008, the Herbicide 2X treatment reduced cover compared to the control, while the Herbicide 1X treatment did not (Figure 1e). Exotic forb cover data for Herbicide 2X are not different from the control or Herbicide 1X in Site 1 (Figure 1b) in 2010. In 2010, however, Site 2 exotic forb cover data for both Herbicide 1X and Herbicide 2X are significantly greater than the control (Figure 1e).

Table 2. Statistics (df, F, and P values) for repeated measures ANOVA and interactions for percent cover at Site 1 (forb site) and Site 2 (grass site). The repeated measure (RM) is cover over three years (2006–2008) with three treatments (herbicide 1X, herbicide 2X, and control) and five replications. Analysis conducted on arcsin transformed data. Data collected June 2006 through June 2010 at Barnett Ranch Reserve, Ramona, California.

Factor		Native Forbs		Exotic Forbs		Exotic Grasses	
Site 1	df	F	P	F	P	F	P
Treatment	2	30.1	<0.0001	16.8	0.0003	11.6	0.0016
RM (cover over 3 years)	2	5.37	0.0118	10.1	0.0007	4.03	0.0309
RM X treatment	4	0.29	0.8795	9.4	<0.0001	10.5	<0.0001
Site 2							
Treatment	2	15.7	0.0004	6.98	0.0098	56.8	<0.0001
RM (cover over 3 years)	2	40.4	<0.0001	65.3	<0.0001	16.9	<0.0001
RM X treatment	4	5.6	0.0024	7	0.0007	12.5	<0.0001

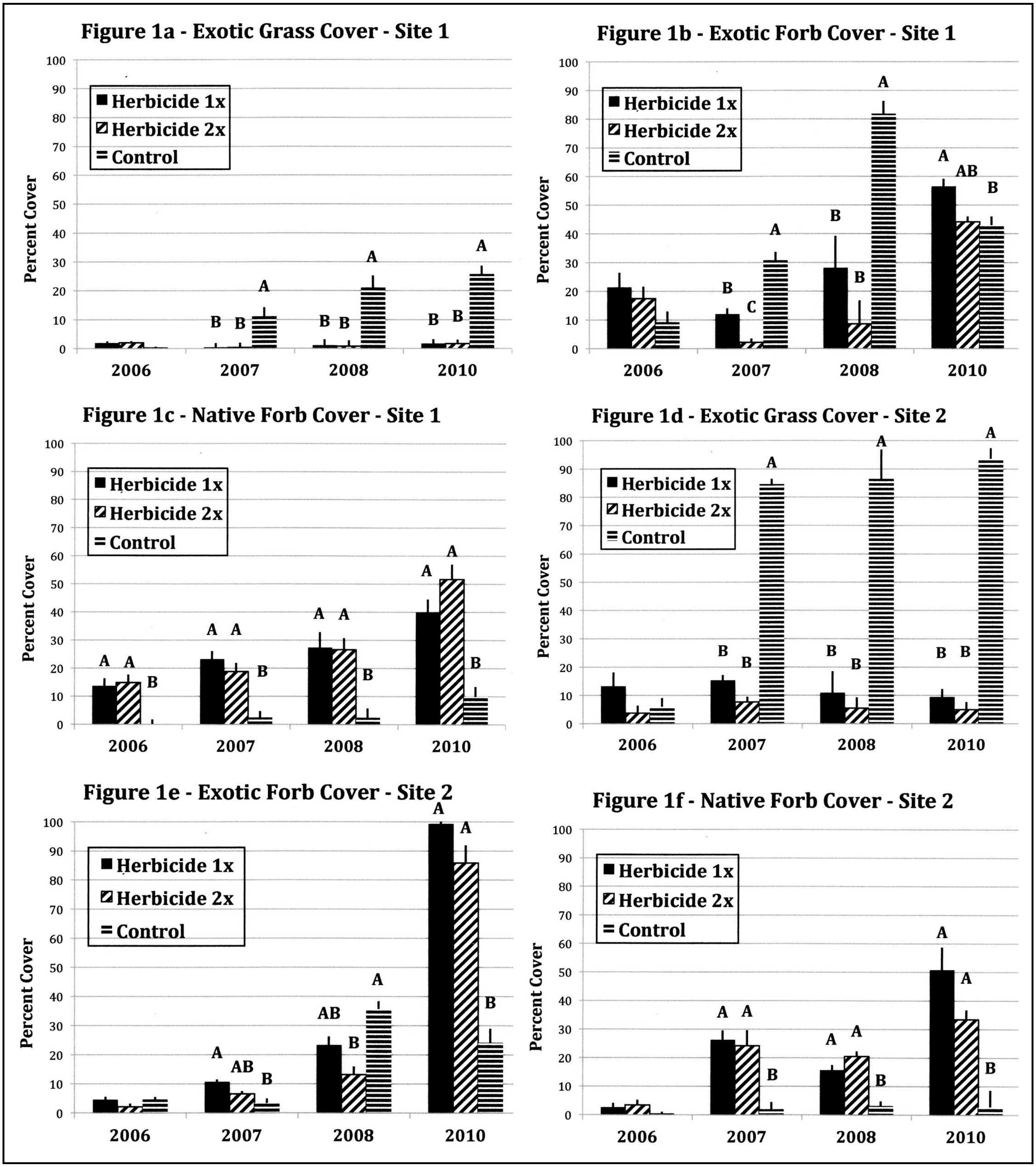

Figure 1. Exotic grass, exotic forb, and native forb percent cover ± SEM by experimental site as affected by treatment, Barnett Ranch Reserve, Ramona, California, 2006 through 2010. Plots were treated with herbicides annually in spring prior to vegetation sampling each May, and were seeded with native species in February 2007. Figure 1a is exotic grass cover in Site 1; 1b is exotic forb cover in Site 1; 1c is native forb cover in Site 1. Figure 1d is exotic grass cover in Site 2; 1e is exotic forb cover in Site 2; 1f is native forb cover in Site 2. Treatment means within a year that share the same letter above the bars are not different according to the Tukey-Kramer Highly Significant Difference Test, $P = 0.05$. Lack of letters above the means indicate that there were no significant differences among treatments.

Native forb cover increased significantly after herbicide treatments in both sites over three years using repeated measures ANOVA, and there was a significant interaction of the repeated measure by treatment in Site 2, but not Site 1 (Table 2). Herbicide treatments improved native forb cover in all three years in Site 1 (F = 5.37 P = 0.012; Figure 1c). In Site 2 there was no significant effect in 2006 (F = 0.93, P = 0.435), but herbicide treatments improved native forb cover in 2007 and 2008 (F = 15.30, P = 0.0018 and F = 18.53, P = 0.001, respectively; Figure 1f). Similarly in 2010, the univariate analyses showed that native forbs had greater cover with either herbicide treatment (F = 17.006, P = 0.0013 for Site 1 and F = 14.435, P = 0.0022 for Site 2). There were no significant differences between the two herbicide treatments in any year for native forb cover at either site, only between the herbicide treatments and the control.

Seeded Native Species

Seeded native species cover was low throughout the experiment. Two of seven seeded species, *M. macrocarpa* and *C. aldersonii*, were not detected in any of the plots in any year. Cover never exceeded 2% for the other five seeded species for the duration of the experiment. Those plants that did emerge, however, were, with one exception, in herbicide treated plots.

Shrub Establishment

The 2011 shrub inventory had enough germinated shrubs to meet the standards of normally distributed data for ANOVA in Site 1 for four species, but not for *S. apiana* (Table 3). Site 2 did not have sufficient shrub data for statistical analysis. In the case of *A. californica*, which was not seeded, small plants (mature plants that survived the 2003 wild fire were not counted) were found in the main plot and subplots regardless of treatment. Although numbers were higher in the herbicide treated subplots (1.9 plants per plot) than the controls (0.4 plants per plot), none of the factors in the model (main plot herbicide treatment, subplot seeding treatment, or the interaction of main plot and subplot) were statistically different. With the three seeded species, main plot treatment was a significant factor in the ANOVA model for *E. fasciculatum* (F = 17.059, P = 0.0004) and the combined inventory for all three species (F = 15.254, P = 0.001), but not for *E. confertiflorum* (F = 2.595, P = 0.120) or *A. glaber* (F = 3.055, P = 0.093). Seeding was a significant factor for all three species (*A. glaber,* F = 11.001, P = 0.003; *E. confertiflorum,* F = 9.133, P = 0.006; and *E. fasciculatum,* F = 37.588, P < 0.0001) and the combined totals (F = 42.253, P < 0.0001). All of the *S. apiana* plants were in herbicide treated plots.

DISCUSSION AND CONCLUSIONS

Native forb and shrub species, both extant and introduced as seed, benefitted from annual herbicide treatments for several years (Figures 1c and 1f, Table 3). The improvement in native species cover seems to be mostly related to controlling exotic annual grasses (Figure 1a vs. 1c and 1d vs. 1f) as was observed in other CSS restoration studies (Cione et al. 2002; Allen et al. 2005; Talluto et al. 2006; Cox and Allen 2008b; Kimball et al. 2014). A total of 43 native species emerged on these two sites with cover up to 50% in the absence of seeding. This is hopeful for restoration efforts that rely entirely on the remnant native seedbank, especially considering that this was a ranch used continuously for more than two centuries.

This also suggests that concentrating on exotic annual grass control can be a useful, simple, low-cost restoration technique (Table 4). The cost of 1X herbicide was estimated at $604/ha total for four years of application. This provided weed control equal to other more costly treatments, and native cover establishment as effectively as more costly seeding. The multiyear approach can also be recommended in order to exhaust exotic seed banks, especially of grass species. However, there is little support in the data for the follow-up herbicide treatment (Herbicide 2X) in terms of weed control and cost compared to just one broadcast application (Herbicide 1X) in the spring (Figure 1, Table 4). Raking litter in the control plots to improve seed contact with soil and hand weed trimming added considerable cost. Plots treated with glyphosate were sparse in litter, providing soil contact for seeds. Hand trimming inflorescences following fluazifop in 2010 did not reduce exotic forbs compared to the herbicide treatments alone (Figure 1b, 1e), so this practice was also not cost-ef-

Table 3. Shrub inventory for three species seeded in Site 1, Barnett Ranch Reserve. Values are mean counts of five plots (5.7 m^2) followed by range of low to high counts per plot. Barnett Ranch Reserve, Ramona, California, June 2011.

Species	Herbicide 1X		Herbicide 2X		Control	
	Seeded	Not Seeded	Seeded	Not Seeded	Seeded	Not Seeded
Acmispon glaber	4.6 (1–9)	2 (0–5)	5.2 (1–13)	1 (0–3)	2.8 (0–8)	0.4 (0–2)
Eriogonum fasciculatum	14.2 (8–25)	1.6 (0–8)	16.2 (12–26)	2.2 (0–6)	0.8 (0–2)	0.2 (0–1)
Eriophyllum confertiflorum	5.2 (0–15)	0	8 (0–19)	0.2 (0–1)	0	0
Combined seeded species	24	3.6	29.4	3.4	3.6	0.6
Artemisia californica (Unseeded)	5.6 (0–16)		2.0 (0–5)		0.8 (0–1)	

Table 4. Field management activity costs, on a per hectare basis in US dollars, March 2006 through April 2011, Barnett Ranch Reserve, Ramona, California.

Treatment	Activity	Materials	Labor	Total
		----------------------- $/ha	-----------------------	
Herbicide 1X	Annual broadcast herbicide application[1]	$354	$250	$604
Herbicide 1X plus seeding	Annual broadcast herbicide application[1]	$354	$250	$2,654
	Seeding[2]	$2,000	$50	
Herbicide 1X plus seeding plus inflorescence trimming	Annual broadcast herbicide application[1]	$354	$250	$6,154
	Seeding[2]	$2,000	$50	
	Inflorescence trimming[3]	0	$3,500	
Herbicide 2X	Annual broadcast herbicide application[1]	$354	$250	$1,376
	Spot spray herbicide[4]	$52	$720	
Herbicide 2X plus seeding	Annual broadcast herbicide application[1]	$354	$250	$3,426
	Spot spray herbicide[4]	$52	$720	
	Seeding[2]	$2,000	$50	
Herbicide 2X plus seeding plus inflorescence trimming	Annual broadcast herbicide application[1]	$354	$250	$6,926
	Seeding[2]	$2,000	$50	
	Spot spray herbicide[4]	$52	$720	
	Inflorescence trimming[3]	0	$3,500	
Control	De-thatch before seeding[5]		$10,000	$12,050
	Seeding[2]	$2,000	$50	

[1] Broadcast herbicide application for 2006–2009; materials were glyphosate at 5 L/ha/year x $13/L x 4 years = $260, fluazifop at 1.3L/one year x $72/ L = $94; labor cost at $20/hour x 2.5 hour/ha/year x 5 years = $250.

[2] Seed cost from suppliers invoice and recommended seeding rates; labor at $20/hour x 2.5 hour/ha for hand seeding.

[3] Inflorescence trimming = utilizing a string trimmer or swing blade, 148 hours/ha x $20/hour in 2010, 49 hours/ha x $20/hour in 2011, applied across all herbicide treated plots.

[4] Spot spray application for 2006–2009; material was glyphosate at 1 L/ha/year x $13/L x 4 years = $52; labor cost $20/hour x 9 hour/year x 4 years = $720.

[5] De-thatch labor cost at $20/hour x 500 hours/ha = $10,000 in first year only.

fective. Manual removal of exotic forb inflorescences to eliminate seed rain was effective for *H. incana* and other bolting exotic forbs (i.e., *Sonchus asper* (L.) Hill ssp. *asper* and *Centaurea melitensis* L.), but did not reduce shorter-statured exotic forb species (i.e., *Erodium* spp., *S. gallica, L. arvensis*) (DiTomaso 2002; Radosevich et al. 2007). This practice was relatively labor intensive, but could be useful in limited situations (Table 4). An added benefit of exotic grass control is reducing the easily ignitable wildfire fuel provided by senesced grasses in landscapes throughout Southern California (Cione et al. 2002; Allen et al. 2005; Bell et al. 2009).

One concern of using a broad-spectrum herbicide, such as glyphosate, is that the native species will also be impacted. Nevertheless, native species cover was significantly greater than untreated plots in all years except the first. This indicates that the timing of our treatments was appropriate to control exotic plants while allowing natives to grow. The phenology of many native species is later than exotic grasses and forbs (Steers and Allen 2010). Even if natives were damaged by herbicide, a sufficient number survived or germinated after herbicide application that they continued to show a positive response to herbicide treatments. Control of exotic forbs (mainly *Erodium* spp.) was successful from 2006 to 2008, but not in the long term in this study. We switched to the grass-specific herbicide fluazifop in 2009 to reduce damage to native forbs, but this also allowed exotic forbs to flourish (Figure 1b). Prior studies indicated that *Erodium cicutarium* (L.) Aiton, and other species of *Erodium,* are controlled by fluazifop (Christopher and Holtum 2000; Steers and Allen 2010), but the dominant species at our site, *E. botrys,* was not controlled. We further attribute the persistence of exotic forbs to their longevity in the seedbank compared to the exotic grass seedbank (Radosevich et al. 2007; Cox and Allen 2008a). Exotic forb cover decreased annually from 2006 through 2008 when treated with herbicide, but the lack of effective control in 2010 apparently allowed the exotic forbs to re-infest the area resulting in very high percent cover in 2010 (Figure 1b and1e). Experiments that compared competition of exotic forbs (especially *Erodium* spp.) with exotic grass (Gillespie and Allen 2004; Cox and Allen 2011) showed that *Erodium* did not inhibit native forb establishment as much as exotic grasses, suggesting control of exotic forbs may not always be critical for restoration. In other words, continuing the annual herbicide application until the exotic forb seedbank was depleted may not be cost effective and may also negatively impact seed production by natives. Furthermore, these were relatively small plots and could easily be reinvaded by adjacent weedy vegetation (Baythavong et al. 2009). All restoration sites are unique and will require adaptive management tailored to the local conditions (Clewell and Rieger 1997).

Our results, as well as published studies on restoration seeding, support two conclusions. One is that seeding was successful in introducing (or re-introducing) some of these species to our study sites (Table 3) (Eliason and Allen 1997; Cione et al. 2002; Cox and Allen 2008b). The other is that eliminating the exotic grasses was clearly beneficial to germination and establishment of these seeded species, as also supported by other studies (Eliason and Allen 1997; Cox and Allen 2008b, 2011; Marushia and Allen 2011; Kimball et al. 2014). Mowing and de-thatching exotic plants was the most expensive and least successful practice, especially when compared to using an herbicide (Table 4) (Allen et al. 2005). Establishment of purchased native seed was less than 0.1% of the seed sown. We do not have an explanation for this result, but lack of sufficient precipitation following sowing is a likely cause (Eliason and Allen 1997; Cione et al. 2002). The expense of the purchased seed clearly outweighs the benefit in this case, and should be weighed in situations where a native seedbank is present.

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LITERATURE CITED

Allen, E.B., J.S. Brown, and M.F. Allen. 2001. Restoration of animal, plant and microbial diversity. Pp. 185-202 *in* Encyclopedia of Biodiversity, Volume 5. Academic Press, San Diego, CA.

Allen, E.B., R.D. Cox, T. Tennant, S.N. Kee, and D.H. Deutschman. 2005. Landscape restoration in southern California forblands: Response of abandoned farmland to invasive annual grass control. Israel Journal of Plant Sciences 53:237-245.

Allen, E.B., S.A. Eliason, V.J. Marquez, G.P. Schultz, N.K. Storms, C.D. Stylinski, T.A. Zink, and M.A. Allen. 2000. What are the limits to restoration of Coastal Sage Scrub in Southern California? Pp. 253-262 *in* 2nd Interface between Ecology and Land Development in California. US Geological Survey, Western Ecological Research Center, Sacramento, CA. Accessed 18 December 2013 <http://pubs.usgs.gov/of/2000/of00-062/of00-062.pdf>.

Baldwin, B.G., D.H. Goldman, D.J. Keil, and R. Patterson. 2012. The Jepson Manual: Vascular Plants of California. University of California Press, Berkeley.

Baythavong, B.S., M.L. Stanton, and K.J. Rice. 2009. Understanding the consequences of seed dispersal in a heterogeneous environment. Ecology 90:2118-2128.

Bell, C.E., J.M. DiTomaso, and M.L. Brooks. 2009. Invasive plants and wildfires in Southern California. UCANR Publication 8397. Accessed 19 December 2013 <http://anrcatalog.ucdavis.edu/pdf/8397.pdf>.

Bell, C.E., J. Ekhoff, and M.Witter. 2013. Herbicides as a tool for rescuing the California State Grass, *Stipa pulchra*. Grasslands 23:5-11.

Burcham, L.T. 1957. California Range Land. California Department of Forestry, Sacramento.

Chen, X., B.L. Li, and M.F. Allen. 2010. Characterizing urbanization, and agricultural and conservation land-use change in Riverside County, California, USA. Ecological Complexity and Sustainability 1195:164-176.

Christopher, J.T., and A.M. Holtum. 2000. Dicotyledons lacking the multi-subunit form of the herbicide-target enzyme acetyl coenzyme A carboxylase may be restricted to the family Geraniaceae. Australian Journal of Plant Physiology 27:845-850.

CIMIS (California Irrigation Management Information System, California Department of Water Resources). 2014. Accessed 4 August 2014 <http://www.ipm.ucdavis.edu/WEATHER/>.

Cione, N.K., P.E. Padgett, and E.B. Allen. 2002. Restoration of a native shrubland impacted by exotic grass, frequent fires, and nitrogen deposition in southern California. Restoration Ecology 10:376-384.

Cleland, E.E., J. Funk, and E.B. Allen. 2016. Coastal Sage Scrub. Pp. 429-448 *in* E. Zavaleta and H. Mooney, eds., Ecosystems of California. University of California Press, Berkeley.

Clewell, A., and J.P. Rieger. 1997. What practitioners need from restoration ecologists. Restoration Ecology 5:350-354.

Cox, R.D., and E.B. Allen. 2008a. Composition of soil seed banks in southern California coastal sage scrub and adjacent exotic grassland. Plant Ecology 198:37-46.

Cox, R.D., and E.B. Allen. 2008b. Stability of exotic annual grasses following restoration efforts in southern California coastal sage scrub. Journal of Applied Ecology 45:495-504.

Cox, R.D., and E.B. Allen. 2011. The roles of exotic grasses and forbs when restoring native species to highly invaded southern California annual grassland. Plant Ecology 212:1699-1707.

Cox, R.D., K.L. Preston, R.F. Johnson, R.A. Minnich, and E.B. Allen. 2014. Influence of landscape-scale variables on vegetation conversion in Southern California, USA. Global Ecology and Conservation 2:190-203.

Dallman, P.R. 1998. Plant life in the world's Mediterranean Climates. University of California Press, Berkeley.

DiTomaso, J.M. 2002. Irrigated pastures. Pp. 331-336 *in* P. Gouveia and F.O. Colbert, eds., Principles of Weed Control, 3rd ed. Thompson Publications, Fresno, CA.

Eliason, S.A., and E.B. Allen. 1997. Exotic grass competition in suppressing native shrubland re-establishment. Restoration Ecology 5:245-255.

Fenn, M.E., E.B. Allen, S.B. Weiss, S. Jovan, L. Geiser, G.S. Tonnesen, R.F. Johnson, L.E. Rao, B.S. Gimeno, F. Yuan, T. Meixner, and A. Bytnerowicz. 2010. Nitrogen critical loads and management alternatives for N-impacted ecosystems in California. Journal of Environmental Management 91:2404-2423.

Fleming, G.M., J.E. Diffendorfer, and P.H. Zedler. 2009. The relative importance of disturbance and exotic-plant abundance in California coastal sage scrub. Ecological Applications 19:2210-2227.

Freudenberger, D.O., B.E. Fish, and J.E. Keeley. 1987. Distribution and stability of grasslands in the Los Angeles Basin. Bulletin, Southern California Academy of Sciences 86(1):13-26.

Gillespie, I.G., and E.B. Allen. 2004. Fire and competition in a southern California grassland: Impacts on the rare forb *Erodium macrophyllum*. Applied Ecology 41:643-652.

Helix Environmental Planning. 2004. Area Specific Management Directives, Barnett Ranch Open Space Preserve. Accessed 27 April 2012 <http://www.sdcounty.ca.gov/reusable_components/images/parks/doc/BarnettRanch_ASMD_2004.pdf>.

Kimball, S., M.E. Lulow, K.A. Mooney, and Q.M. Sorenson. 2014. Establishment and management of native functional groups in restoration. Restoration Ecology 22:81-88.

Marushia, R.G., and E.B. Allen. 2011. Control of exotic annual grasses to restore native forbs in abandoned agricultural land. Restoration Ecology 19:45-54.

Minnich, R.A., and R.J. Dezzani. 1998. Historical decline of coastal sage scrub in the Riverside-Perris Plain, California. Western Birds 29:366-391.

Noss, R.F., E.T. Laroei, J.M. Scott. 1995. Endangered ecosystems of the United States: A preliminary assessment of loss and degradation. Biological Report 28, US National Biological Service, Washington, DC. Accessed 25 April 2012 <http://noss.cos.ucf.edu/papers/Noss%20et%20al%201995.pdf>.

Radosevich, S.R., J.S. Holt, and C.M. Ghersa. 2007. Ecology of Weeds and Invasive Plants. Wiley-Interscience, Hoboken, NJ.

Rundel, R.W. 2007. Sage Scrub. Pp. 208-228 *in* G. Barbour and J. Majors, eds., Terrestrial Vegetation of California. Wiley and Sons, New York.

Scott, T.A., L. Fernandez, and M.F. Allen. 2006. Land use planning. Pp. 206-217 *in* J.A. Scott, D.D. Goble, and F.W. Davis, eds., The Endangered Species Act at Thirty. Volume 2: Conserving Biodiversity in Human-Dominated Landscapes. Island Press, Washington, DC.

Steel, R.G.D., and J.H. Torre. 1980. Principles and Procedures of Statistics. McGraw-Hill, New York.

Steers, R.J., and E.B. Allen. 2010. Post-fire control of invasive plants promotes native succession in a burned desert shrubland. Restoration Ecology 18:334-343.

Stylinski, C.D., and E.B Allen. 1999. Lack of native species recovery following severe exotic disturbance in southern California shrubland. Journal of Applied Ecology 36:544-554.

Talluto, M.V. and K.N. Suding. 2008. Historical change in coastal sage scrub in southern California in relation to fire frequency and air pollution. Landscape Ecology 23:803-815.

Talluto, M.V., K.N. Suding, and P.A. Bowler. 2006. Factors affecting understory establishment in coastal sage scrub restoration. Madroño 53:55-59.

Weathers, K.A. 2013. The role of invasive *Erodium* species in restoration of Coastal Sage Scrub communities and techniques for control. PhD dissertation, University of California, Riverside.

Weaver, J.E., and F.E. Clements. 1938. Plant Ecology. McGraw-Hill, New York.

Westman W.E. 1981. Diversity relations and succession in Californian coastal sage scrub. Ecology 62(1):170-184.