THESIS

USING NATIVE ANNUAL PLANTS TO SUPPRESS WEEDY INVASIVE SPECIES IN POST-FIRE HABITATS

Submitted by

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WE HEREBY RECOMMEND THAT THE THESIS PREPARED UNDER OUR SUPERVISION BY CHRISTOPHER M. HERRON ENTITLED USING NATIVE ANNUAL PLANTS TO SUPPRESS WEEDY INVASIVES IN POST-FIRE HABITATS BE ACCEPTED AS FULFILLING IN PART REQUIREMENTS FOR THE DEGREE OF MASTER OF SCIENCE.

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ABSTRACT OF THESIS

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Increasing fire frequencies and uncharacteristic severe fires have created a need for improved restoration methods across rangelands in western North America. Traditional restoration seed mixtures of perennial mid- to late-seral plant species may not be suitable for intensely burned sites that have been returned to an early-seral condition. Under such conditions native annual plant species are likely to be more successful at competing with exotic annual plant species such as *Bromus tectorum* L. We used a field study in Colorado and Idaho, USA to test the hypothesis that native annual plant species are better suited to post-fire restoration efforts compared to perennial plant species that are commonly used in traditional seed mixtures. Replicated test plots at four post-fire sites were assigned one of four treatments (1) native annual seed mixture, (2) standard perennial seed mixture, (3) combination of annual and perennial, and (4) an unseeded control. Results suggest that there is potential for native annual plant species to be effective competitors with weedy exotic species in post-fire restoration scenarios.

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INTRODUCTION

Wildland fires in western North America are becoming more frequent and are expected to increase in frequency with the current trend in climate change (Westerling et al. 2006). However, fires have historically been a common element of western North American landscapes with fire return intervals of 60-100 years in some rangeland communities (Whisenant 1990) and decades-centuries in piñon-juniper communities (Floyd et al. 2004). However, ecosystems throughout the west have experienced displacement of native vegetation by exotic invasive annuals like *Bromus tectorum* L., which reduces fire return intervals to 3-5 years (Whisenant 1990). *B. tectorum* is potentially able to displace native vegetation after a fire due to its rapid growth during fall and spring (Harris 1967), its ability to germinate in a wide array of environmental conditions (Mack and Pyke 1983) and its ability to utilize soil resources before native plants are able to (Melgoza et al. 1990). With more frequent fires and the increased potential for cheatgrass invasion, restoration is an important practice during the initial post-fire period.

An immediate goal of post-fire rehabilitation has been stabilization to prevent loss of soil and soil productivity (Robichaud et al. 2000). Fire rehabilitation projects have included species that are able to protect watersheds and provide forage but they have not taken into account the long-term effects on ecosystem dynamics (Richards et al. 1998). In the western U.S., exotics such as crested wheatgrass (*Agropyron cristatum* (L.) Gaertn.), Siberian wheatgrass (*Agropyron fragile* [Roth] Candargy), and alfalfa (*Medicago sativa* L.) have been used extensively to provide initial post-fire vegetative cover (Richards et al. 1998). However, this practice has resulted in exotic-dominated plant communities that can hinder native plant regeneration (Chambers et al. 1994;

Walker 1997; Kruse et al. 2004). The likelihood of accidentally seeding exotics may also be greater when seeding large areas (Robichaud et al. 2000, Hunter and Omi 2006, Keeley et al. 2006). These issues compound the management problem of native vegetation recovery in post-fire habitats.

Typically, restoration seed mixtures have been comprised of late seral or climax species and sometimes non-native species (Lesica and DeLuca 1996; Pellant and Monsen 1993; Beyers 2004). Restoration seed mixes like these can be insufficient for preventing weedy plant species from establishing and persisting on post-fire landscapes (Beyers 2004; Floyd et al. 2006). However, the use of native species in seed mixes is currently increasing (Thompson et al. 2006; Dorner 2002) as land managers take into consideration changing social values (Johnson 1986), shifts in policies (Hoberg 1997), and advances in ecological knowledge (Roundy et al. 1995).

Planting perennial species in conditions where annuals typically would dominate may be tempting from a revegetation perspective, but may not be the best choice in post-fire restoration settings. Kardol et al. (2006) and Kulmatiski (2006) have both shown that later-seral species perform poorly in early-seral soils. Young and Evans (1978) found that post-fire soils in sagebrush grasslands had accumulated relatively large amounts of soil nitrogen. A variety of studies have shown that annual plant species tend to perform better than perennial species under high nitrogen conditions. Tilman (1987) described plant community dynamics of old fields in Minnesota through a study of nutrient addition and varied successional stages. He found annual plant species initially increased with nutrient additions while the perennial species decreased in abundance until several years had passed and the perennials were able to reestablish (Tilman 1987). Another study in the tall grass prairie found that the first year after nutrient manipulations early seral species dominated the study plots and during subsequent years they gave way to native perennial species (Baer et al 2005). During a study of nitrogen addition and immobilization in the short-grass steppe of Colorado, Paschke and others (2000) found that increased soil nitrogen resulted in increased biomass of annual plant species while immobilization of soil nitrogen through sucrose additions resulted in increased biomass of perennial plant species. Annual plant species are clearly favored under conditions of increased soil nitrogen while perennial species have been shown to be reduced.

After intense disturbances, pioneer or ruderal species, the majority of which would be annuals followed by short-lived perennials and/or biennials, would typically establish first and dominate the first year or two after the disturbance (Borgegard and Rydin 1989, Bazzaz 1996, Walker and Moral 2003, Ott et al. 2003, Keeley et al. 2006). Everett and Ward (1984) found that annuals can dominate early post-fire vegetation in Piñon- juniper ecosystems in eastern Nevada, even if they were sparse pre-fire. Shinneman and Baker (2009) also found that there was an initial dominance of annual and perennial forbs in the first 1-2 years post-fire, which then gave way to perennial grasses in a western Colorado Piñon-juniper ecosystem. Koniak and Everett (1982) found annual species comprised 89% of emerging seedlings in seedbanks of six successional stages of Piñon-juniper woodlands in California, while there were few perennial species present in the same seedbanks.

However, a number of researchers (Young et al. 1969; Young and Evans 1978; Melgoza et al. 1990; Evangelista et al. 2004; Jessop and Anderson 2007) have found that instead of native annuals colonizing and dominating post-fire habitats (in a variety of habitat types throughout the west), there is an increasing trend toward exotic annual species (especially *B. tectorum*) colonizing and then dominating sites. This could be due to a close proximity to an established exotic species population, close proximity to dispersal vectors such as roadways, the alteration of the historical fire regime (Keeley et al. 2005), or a lack of local native annual seed in the seed bank. Native annual seed in seed banks could be destroyed because of the increasing frequency and intensity of fires in the western U.S. (Westerling et al. 2006), which in turn leaves empty niche space for exotic invaders to colonize and dominate.

When *B. tectorum* colonizes, it can accumulate enough fine fuels to carry fire between interspaces (Young and Evans, 1978) and also increase the probability, frequency, and intensity of fires (Knapp, 1996), which makes native plant recovery difficult. To combat these issues, revegetation efforts may do well by establishing species that compete with *B. tectorum* during the time of resource use (Booth et al. 2003), in addition to species that stabilize soils and are otherwise desirable.

Planting annuals has been indicated as a possible means to combat *B. tectorum* (Whisenant, 1990). Because annuals have similar growth forms and phenology as *B. tectorum*, they can be successful competitors for soil resources (Chambers et al. 2007), they can act as a cover crop in early growth stages to compete for light resources (Perry et al. 2009), and they can be the foundation for successional management and ecologically based invasive plant management (Krueger-Mangold and Sheley, 2006). Whisenant (1990) stated that ruderal or pioneer species may accelerate successional processes by (1) stabilizing the soil; (2) increasing soil organic matter; (3) enriching soil nutrients; and (4) excluding less desirable pioneer species through competition.

Because of their known important ecological roles, I hypothesized that native annual plant species would be a better match for post-burn sites than the commonly planted perennial species and would thus provide better initial plant cover (H1). I also hypothesized that these native annual species would be superior competitors with exotic annual species and would thus result in reduced cover of exotic annuals (H2)

METHODS

Sites

A field study was conducted at four separate sites (Table 1), two each in Idaho and Colorado. One study site in Idaho was located within the Craters of the Moon National Monument & Preserve (hereafter referred to as Craters) and the second study site in Idaho was located west of the city of Twin Falls. One Colorado study site was located south of the town of Dinosaur while the second Colorado study site was located north of the town of DeBeque. Each study site was selected because it had recently burned (summer-fall of 2007), it was within close proximity to an existing population of *B. tectorum*, and was an area where land managers were concerned about post-fire invasion of *B. tectorum*. Each site has historically been subjected to grazing by livestock and wildlife. In the spring of 2008 a welded-wire fence was installed at each site to prevent grazing of the study plots by livestock.

Table 1. Characteristics of the four field sites used in this study. Habitat, historic climate, and soil information are from: Soil Survey Staff, NRCS (2009).

Site	Fire	Habitat Type	Precip. (mm.)	Temp. (°C)	Soils
Craters, ID	Bear Den Butte	Basin big sagebrush/Bluebunch wheatgrass	35-40	7-10	Loamy-skeletal, mixed, superactive, mesic Calciargidic Argixerolls
Twin Falls, ID	Murphy Complex	Basin big sage/Bluebunch wheatgrass – Thurber's needlgrass	25-30	8-10	Loamy, mixed, mesic, shallow Xerollic Durargids
Dinosaur, CO	Steuwe	Piñon-Juniper	25-30	6-8	Fine-loamy, mixed Borollic Haplargids
DeBeque, CO	Pyramid	Piñon-Juniper	25-30	8-11	Fine-loamy, mixed, mesic Ustollic Natrargids

Experimental Design

The study was a randomized complete block design. Each of the four sites contained seven blocks with four treatment plots randomly assigned in each block. Treatments were established in 2- x 2-m plots. The treatments were (1) a native annual seed mixture, (2) a native perennial seed mixture, (3) a combination of the annual and perennial mixtures, and (4) an unseeded control (Table 2). Seed mixtures (1) and (2) were seeded at a rate of 650 PLS m^{-2} and mixture (3) was seeded at a rate of 1300 PLSm^{-2}. The higher rate for mixture (3) was based on the assumption that annual species would drop out of the community within a few years.

Table 2. Species seeded in Native Annual treatment (1) and Perennial treatment (2); the combination/mix seed treatment (3) used all species listed at a combined rate. Seeding rates are represented as Pure Live Seeds per square meter.

Seed Mix	Scientific name	Common name	PLS m^{-2}
1. Native Annual	*Amaranthus retroflexus* L.	Redroot amaranth	78
	Cleome serrulata Pursh.	Rocky Mountain bee-plant	65
	Coreopsis tinctoria Nutt.	Golden tickseed	78
	Helianthus annuus L.	Annual sunflower	65
	Verbena bracteata Cav. ex Lag. & Rodr.	Big bract verbena	65
	Aristida purpurea Nutt.	Purple three-awn	39
	Vulpia microstachys (Nutt.) Munro	Small fescue	130
	Vulpia octaflora (Walter) Rydb.	Six-weeks fescue	130
2. Native Perennial	*Balsamorhiza sagitatta* (Pursh) Nutt.	Arrowleaf balsamroot	84.5
	Eriogonum umbellatum Torr.	Sulfur-flower buckwheat	97.5
	Oenothera pallida Lindl.	Pale evening primrose	52
	Sphaeralcea munroana (Douglas) Spach.	Munro's globemallow	84.5
	Achnatherum hymenoides (Roem. & Schult.) Barkworth	Indian ricegrass	65
	Elymus elymoides (Raf.) Swezey	squirreltail	58.5
	Elymus lanceolatus (Scribn. & J.G. Sm.) Gould	Thickspike wheatgrass	71.5
	Pascopyrum smithii (Rydb.) A. Löve	Western wheatgrass	58.5
	Pseudoroegneria spicata (Pursh.) A. Löve	Bluebunch wheatgrass	78

The Idaho plots were established on October 30th & 31st 2007 and the Colorado plots were established on November 13th and 14th 2007. After the study plots were located and marked, but before treatments were applied, soil cores were taken to be used for a seedbank study. Treatment plots were prepared by raking with a leaf rake to remove debris followed by raking with a garden rake to prepare the seedbed. Seeds were then hand broadcast onto the plots, which were lightly raked by hand to incorporate the seed. We were concerned that there would be insufficient *B. tectorum* seeds remaining on the soil surface following the burns. Therefore, material that was removed from the plots with the initial raking was pooled at each site and to this material litter from interspersed unburned *B. tectorum* patches was collected and added. This site specific pooled material was mixed and then an equal volume was added to each plot in order to ensure more uniform *B. tectorum* propagule supply. After this material was added, each plot was rolled with a water-filled lawn roller. This was done to firm the seedbed and to reduce wind erosion of the seed and raked soil.

In order to effectively test my hypotheses I required the moisture at each site to be at least equal to the average annual precipitation levels. Therefore, precipitation levels were monitored using the High Plains Regional Climate Center observation stations (High Plains Regional Climate Center, 2010, see http://www.hprcc.unl.edu, accessed throughout study) nearest each site. During the fall 2007 – spring 2008 timeframe, the Idaho sites had fallen below the 30 year average annual level of precipitation (High Plains Regional Climate Center, 2008, station ID# 102260). To help offset this deficit; the equivalent of an additional centimeter of precipitation (one time only) was added to each plot at the Idaho sites in May of 2008. Water applications were made by hand with garden watering cans. During fall 2008 – spring 2009 timeframe, all four sites had received at least their average annual precipitation (High Plains Regional Climate Center, 2009, Station ID #'s 102260, 101551, 056832, 056266).

Seedbank Analysis

A 5-cm diameter bulb planter was used to collect two soil samples per plot to a depth of 5 cm, resulting in eight samples per block. The eight samples for each block were combined to be a representative composite sample of the individual blocks at each study site. These samples were used to determine post-fire seedbank composition. The soils were subjected to a cold stratification process by storing in a refrigerator for 16 weeks at 5 °C, before being transferred to the greenhouse at Colorado State University where they were maintained at 18 – 21 °C with supplemental lights that maintained a photoperiod of 16 hours. Half of each composite sample was spread onto a mixture of 2/3 Fafard™ Superfine Germination Mix potting soil and 1/3 play sand in a growth flat (26.7 cm x 53 cm) and placed onto heating pads set to 24.0°C to improve germination. Flats were watered as needed when samples were dry and randomized on a weekly basis. As plant species germinated they were identified by species when possible, and counted to calculate seed density in the seedbank (# germinating seedlings divided by the 0.0078 m^2 cross-sectional area sampled).

Sampling and Data Collection

Data collection occurred during May of 2008 and 2009 to capture as much of the ephemeral annual cover as possible during the spring timeframe. We measured percent cover by species and counted the number of individuals of seeded species in each plot. The measurements were made in four permanent 0.1875-m^2 subplots within each 4-m^2 plot. Comparing these measurements of total plant cover, exotic annual cover, and native perennial cover between treatments allowed us to test each of the hypotheses. Counting the number of individuals of seeded species resulted in densities of those species that were used to determine seeding success and evaluate potential competitive ability.

Statistical Analysis

SAS™ 9.2 was used to analyze data. All analyses were performed at $\alpha=0.05$. Variables were square root transformed to meet assumptions of the analysis. All comparisons were evaluated using a mixed effects model. Means comparisons were made by looking at the differences of the least square means between treatments and adjustments from Tukey's multiple comparisons. Initially, treatment was held as a fixed effect to evaluate possible broad scale differences between treatments across all sites and years. Treatment by year interactions were also investigated. A fine scale analysis held site and treatment as fixed effects to examine the differences between treatments at each site. Treatment by year interactions were also investigated at this scale to examine annual variability of exotic annual and total plant cover between treatments at each site.

Linear regressions using Pearson's R-values were used to determine if any variables were significantly related ($p<0.05$) to reduced *B. tectorum* and exotic annual cover. Regression analysis was performed on sampling years 2008 & 2009 pooled together to determine if there were any broad scale correlations, each year separately to examine differences in years, each site with years pooled to look at site specific effects, and site by each sampling year to look at the variations between sites and years.

RESULTS

Seedbank study

The Craters, ID seedbank had two species germinate with only one individual per species. The Twin Falls, ID seedbank had two species germinate with four individuals. The Colorado seedbanks, Dinosaur and DeBeque, had greater diversity with nine and seven species, respectively, and between one and twelve individuals for each species (Table 3). Some collected species did not reach maturity to be identified during the timeframe of the study and were left as unknowns.

Table 3. Species germinated during seedbank study and the estimated densities (# m^{-2}) (±se) of each in the seedbanks at their respective sites.

Species	Craters	Murphy	Dinosaur	DeBeque
Bromus tectorum	na	na	805.9 (210.7)	18.3 (18.3)
Vulpia octoflora	na	na	128.2 (62.3)	36.6 (23.6)
Sysimbrium altissiumum	na	na	109.9 (76.3)	na
Plantago patagonica	na	na	18.3 (18.3)	na
Alyssum parviflorum	na	na	18.3 (18.3)	na
Oenothera pallida	na	na	na	18.3 (18.3)
Descurania pinnata	na	54.9 (25.9)	na	36.6 (23.6)
Unknown forbs	36.6 (23.6)	18.3 (18.3)	73.3 (54.9)	54.9 (38.1)

Plant cover

Treatment effects were seen with total plant cover when treatment was the only fixed effect. The annual treatment had significantly less total plant cover compared to the control treatment ($p=0.0467$), there were no other significant differences between treatments at this level. At the treatment by year level, 2009 yielded significantly less total plant cover in the annual treatment compared to the control ($p=0.0036$) and perennial (p=0.0026) treatments. There were no differences between treatments in 2008. Exotic annual cover did not show any treatment effects at these scales.

When treatment and site were held as fixed effects there were no differences in total plant cover between treatments at any site. There was a difference of exotic annual cover at the DeBeque site. The native annual treatment had significantly less exotic annual cover compared to the control (p=0.0328). Total plant cover and exotic annual cover both exhibited differences between treatments at the treatment by year level. At the DeBeque site during 2009, there was significantly less total plant cover in the annual treatment compared to the control (p=0.0521) and perennial (p=0.0471) treatments. At the Dinosaur site during 2009, there was significantly less exotic annual cover in the annual treatment compared to the perennial treatment (p=0.0179). Linear regression indicated that there were slight negative correlations between native annual forb cover and *B. tectorum* cover (R^2= 0.28, p=<0.0001) as well as between native annual forb cover and exotic annual cover (R^2= 0.21, p=0.0004) during 2009 at the Dinosaur site.

DISCUSSION

Site differences

Each site had unique environmental conditions (soils, vegetation, climate and fire effects), which likely resulted in distinct responses to the seeding treatments. Soils and vegetation were the primary differences. The Craters site pre-burn vegetation was late-seral *Artemisia tridentata* Nutt. *ssp tridentata – Pseudoroegnaria spicata* (Pursh.) A. Löve with a suite of other species that are characteristic of late-seral sagebrush steppe habitats (based on our observations of surrounding unburned vegetation). The Bear Den Butte fire that burned through this site was described as hot with severe behavior and spotting (Sammi 2007). Our study site was located near an edge of the burn and was capable of receiving seed rain from surviving vegetation. We also observed re-sprouting of the bunch grasses and other perennial vegetation at this study site. The Twin Falls study site pre-burn vegetation was *Agropyron cristatum* (L.) Gaertn. with a *B. tectorum* component**.** The Murphy complex fire that burned this site was hot and very fast, overtaking livestock on the range (BLM 2009). There were remnant *Agropyron cristatum* root crowns that survived the fire and re-sprouted. Both of the Colorado sites, Dinosaur (Steuwe fire) and DeBeque (Pyramid fire) were within the piñon-juniper habitat type (Soil Survey Staff, NRCS 2009). Both of the Colorado fires were small ground fires that were lightning ignited. However, the soils were different at the two sites (Table 1). The Dinosaur site had loamy soils with a clay component while the DeBeque site had very fine sandy loam soils.

The seedbanks and the initial plant assemblages may help explain some of the treatment effect variability observed between sites. The Idaho seedbanks appeared depauperate and each site exhibited a strong presence of perennial vegetation in the initial plant assemblage. In contrast, the Colorado seedbanks expressed greater diversity but the perennial vegetation was not as well represented in the post-fire community. The Idaho sites had reduced cover of exotic annuals across all treatments (relative to the Colorado sites) but treatment effects could not be discerned. The Colorado sites had much greater exotic annual cover but treatment effects were more prominent at these sites. The stunted seedbanks of the Idaho sites may not have contained enough exotic annual seed to accurately test our hypotheses and the rapid recovery of perennial vegetation may have skewed our results. The intact seedbanks of the Colorado sites provided ample exotic annual seed to test our hypotheses and the initial vegetation lacked a strong perennial component that may have influenced the results.

These four sites represent a small amount of habitat diversity across the Rocky Mountain west but have allowed me to test my hypotheses under variable conditions to provide an idea of how each treatment might perform under similar conditions in a management setting.

Seedbank study

The Idaho sites had the lowest species richness and densities of seeds post-fire. Neither Idaho seedbank contained any *B. tectorum* (Table 2). If the Idaho fires were intense enough, *B. tectorum* seeds could have been consumed or killed (Keeley and McGinnis 2007) as would any other seeds that were on or near the soil surface. The Colorado sites experienced low-severity ground fires and the seedbank samples contained viable *B. tectorum* seeds and had greater species richness and densities than the Idaho seedbanks. A study of varying fire severities on native grass seedbanks (Hunter and Omi 2006) found that low severity fires resulted in significantly more native grass establishment. The Colorado seedbanks yielded a native grass as well as native forbs (Table 2).

Using native annual plant species for improved initial plant cover (H1)

There were no results to indicate that using native annual plant species would provide improved initial plant cover. However, during the analysis, when treatment was the only fixed effect there was significantly less total plant cover in the annual treatment compared to the control treatment. Estimates of exotic annual cover indicated that there was reduced cover of exotics in the annual treatment compared to the control treatment ($t = -3.04$). Although there was not a significant difference of exotic annual cover between treatments at that scale, the estimates may explain why there was less total plant cover in the annual treatments compared to the controls. If there was reduced exotic annual cover in the annual treatments, there would also be reduced total plant cover in those treatments as well. This would support the second hypothesis.

Using native annual plant species to suppress exotic annual plant species (H2)

The Colorado sites had results that supported the hypothesis that native annuals could suppress exotic annuals. The primary exotic annual plant species that were present at both Colorado sites was *B. tectorum*; the secondary species were exotic annual mustards (*Sisymbrium altissimum* L. & *Descurania sophia* (L.) Webb ex Prantl). The DeBeque site had significantly reduced exotic annual cover in the native annual treatment compared to the control, while the perennial treatment did not. At the Dinosaur site there was significantly reduced exotic annual cover in the annual treatment compared to the perennial treatment during 2009.

The Idaho sites did not support this hypothesis. Both sites showed much less exotic annual cover compared to the Colorado sites. This could be due to a high level of native perennial grass that survived the Bear Den Butte fire at the Craters site and conversely, the high level of exotic perennial grass (*A. cristatum)* that survived the Murphy Complex fire at the Twin Falls site.

Although both Idaho sites had greater relative native cover compared to the Colorado sites (Fig. A2), the treatment effects were only present at the Colorado sites. It is possible that the sampling methods were not able to capture the full spectrum of native annuals that may have emerged to compete with the exotic annuals. It is possible that there were flushes of the seeded native annual species that responded to variability in precipitation throughout the remainder of the summer and fall growing seasons. There were senesced skeletons of *Helianthus annuus* and *Cleome serrulata*, which were both seeded as part of the native annual seed treatments that appeared to be from the previous growing season – based on stem color and brittleness. In addition, a linear regression analysis indicated that native annual forbs at DeBeque, during 2009, had a weak but significant negative correlation with *B. tectorum* (R^2= 0.21, p=0.01). Further, the density of *Helianthus annuus* also had a weak but significant negative correlation with exotic annuals (R^2= 0.18, p=0.02) at the Dinosaur site during 2009. These large native annual forbs could have been capable of effectively competing for light and soil resources. Previous research has indicated that annual forb species may be effective competitors with *B. tectorum* for these resources (Young and Evans 1978; Pokorny et al 2005; Perry et al. 2009). Furthermore, at the DeBeque site (2009) (Fig. A2), it appeared that when there was increased cover of exotic annual forbs there was decreased cover of exotic annual grass (R^2= 0.37, p=0.0006). This supports the idea that annual forb species, exotic or native, could successfully compete with exotic annual grass species such as *B. tectorum*. Although it appears that the native annual grasses played little to no role in competing with *B. tectorum* and other exotic annual species, the native annual grasses could be responsible for added competitive pressure during critical developmental stages of *B. tectorum*.

MANAGEMENT CONSIDERATIONS

Seeding post-fire habitats with native annual species might be an effective management approach because of their tendency to grow in these early-seral conditions and for their ability to compete with exotic annuals at the phenological level (Chambers et al 2007). A major limitation of this management approach is the lack of native annual species that are commercially available from seed suppliers. Since there is little to no market for these species currently, it is not likely that a seed company will invest in producing them. However, due to the tendency of annuals to invest heavily in seed production (Smith et al 2010), commercially available native annual species might be easily produced. Another advantage of the annual species for restoration efforts is that they typically have broadly adapted genotypes with broad ranges (Bazazz 1996). This means that local adaptations for a particular species' ecotype are less of a concern. Furthermore, projects that have used ruderal annual species, especially forbs, in restoration efforts have shown superior establishment success (Smith et al 2010; Pywell et al. 2003).

While the idea of using native weedy species to combat exotic weeds is not a novel concept, I believe that the idea of using native annual species to combat exotic annuals in post-fire arid and semi-arid habitats represents a unique and promising management approach. At the Colorado sites the native annual seed treatment had noticeable effects on exotic annual cover. Based on these results, a simple addition of native annual plant species would be an easy modification to common perennial seed mixtures and may provide some competition for exotic annual plants in post-fire habitats. As indicated by Brown (2004), including multiple functional guilds in restoration seed mixtures may increase community competitiveness and provide a buffer against non-native plant invasions. However, the seeding treatments may be less effective if wind and soil movement influence the treatment application.

There should be more research on the use of native annual forbs due to their ability to effectively compete with exotic annuals, especially *B. tectorum*. Native annual grasses should not be overlooked either and should be researched further with grass species that are more robust than the two *Vulpia* species that were used in this study. Also due to the varying success of the native annual species between sites there should be more research with these native annual species and specific post-fire soil conditions, specifically nitrogen availability, to determine how varying fire severities may influence the success of native annual seed treatments.

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APPENDIX

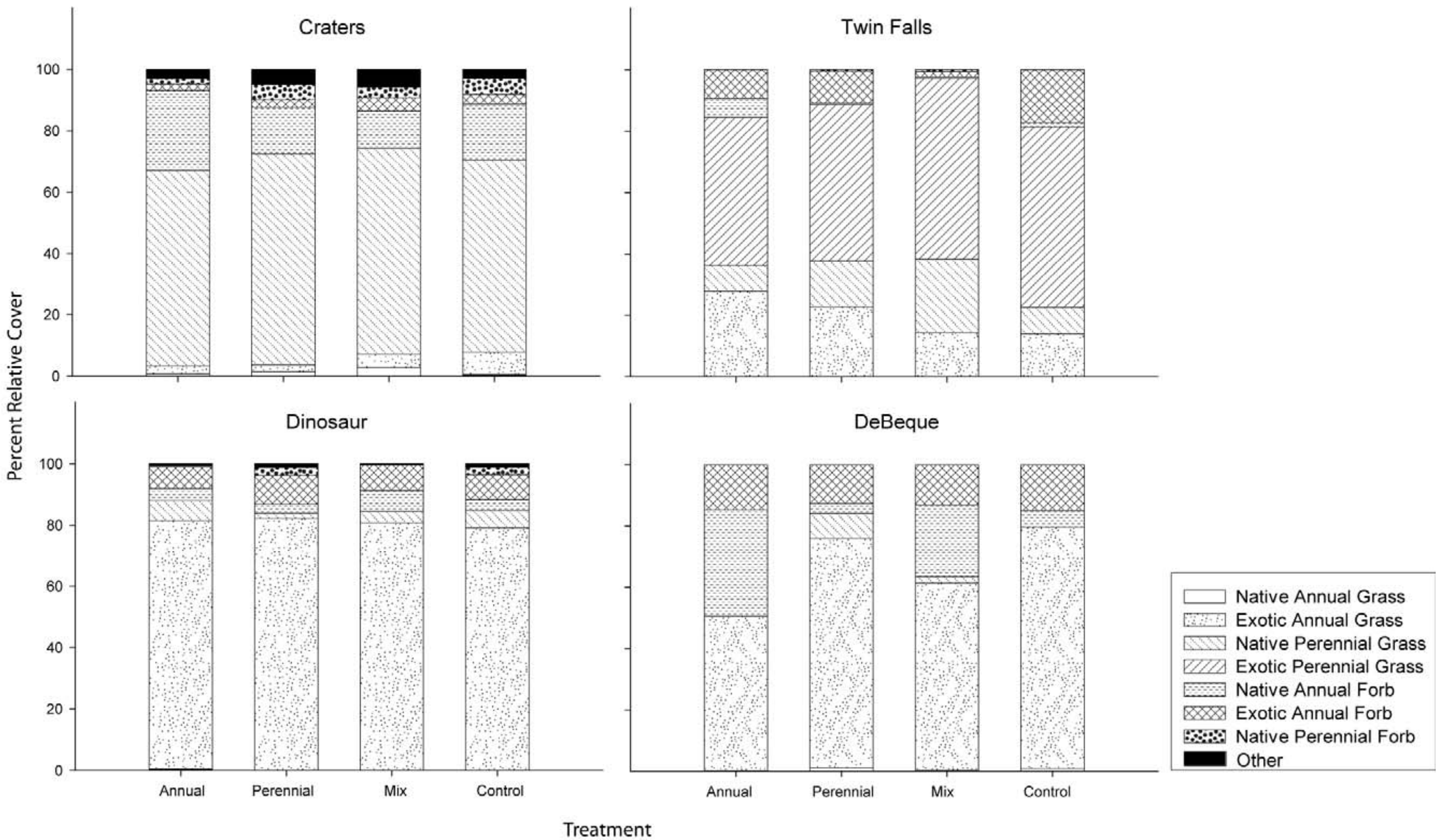

Figure A1. Relative native and exotic plant life forms by site and treatment for sampling year 2008, variable "other" (black bars at top) includes shrub, lichen, sedge, and unidentified species.

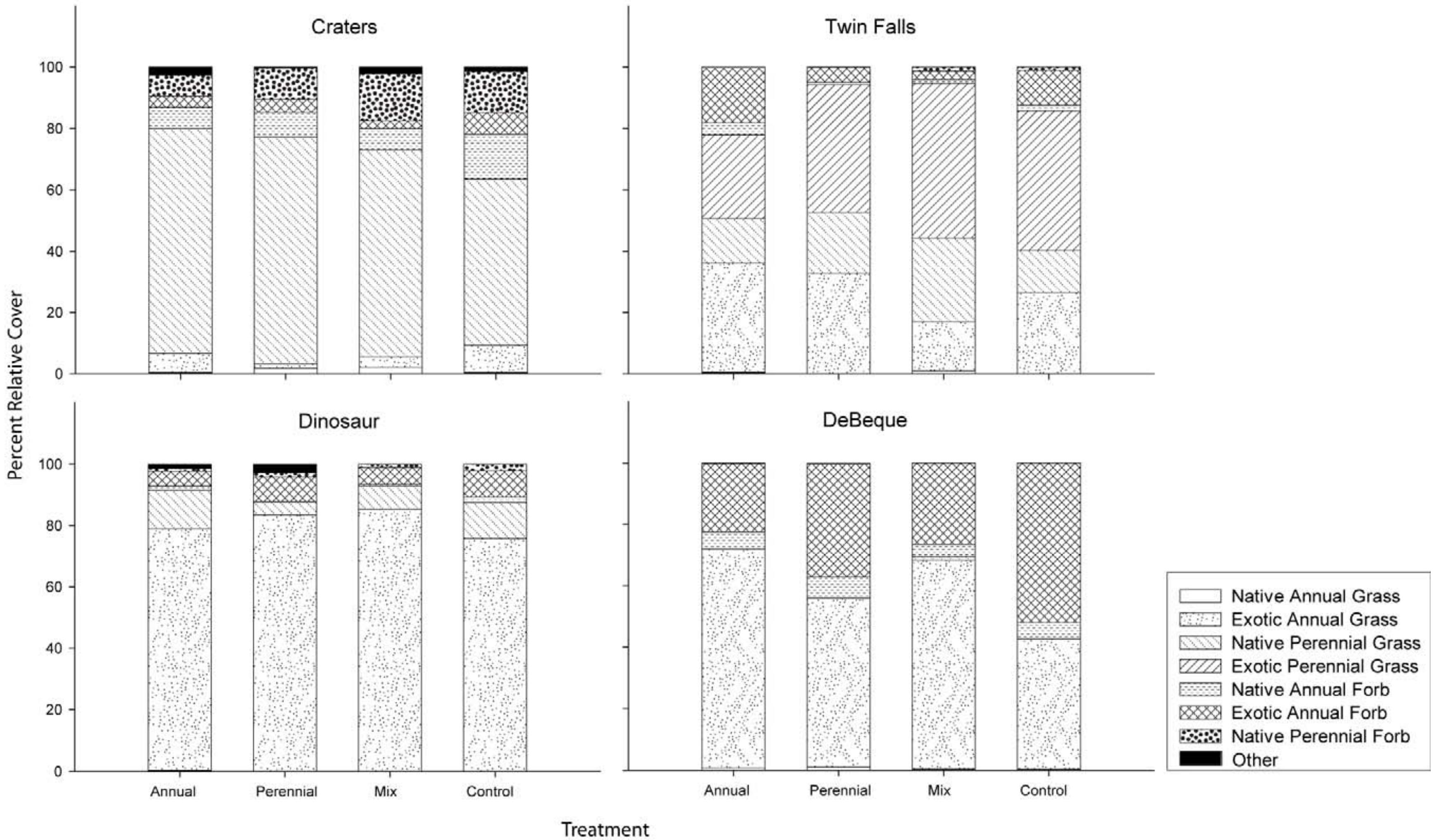

Figure A2. Relative native and exotic plant life forms by site and treatment for sampling year 2009, variable "other" (black bars at top) includes shrub, lichen, sedge, and unidentified species