Project Title:
Tending the land: Integrating indigenous knowledge into the restoration of a California grassland

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Executive Summary

This project took an integrated approach to grassland restoration at Pinnacles National Park (PNP), by incorporating traditional ecological knowledge (TEK) with contemporary techniques to restore and protect the natural and cultural processes of a California grassland system. Along with our academic and tribal partners, the Amah Mutsun Tribal Band and Chalon people, we expanded on an ongoing grassland restoration project by incorporating additional plant species that have both cultural value and are ecologically important to California grasslands. The project was designed around two objectives: 1) continue sampling of plant community characteristics, deergrass plant and inflorescence density, and rhizome sampling of white root sedge within the research plots in McCabe Canyon that were established in 2009, and 2) examine the effects of mowing, topsoil removal through scraping, broadleaf herbicide applications, and seed additions on restoring degraded grasslands within the bottomlands at Pinnacles National Park. The project builds upon an extensive and ongoing research-based framework that aims to evaluate and incorporate indigenous management techniques, while also enhancing native biodiversity and restoring culturally important plants.

Based on our findings we found that our treatments of Carex barbarae did not respond to traditional tending practices at our site with rhizomes of the quality needed for use in basketry. Comments from our tribal practitioners suggested that this may be because the soils at our site had too much clay and silt, and that tending may not have been frequent enough. Therefore, we recommend that in future efforts, Carex barbarae be tended annually and in sandy soils to promote the conditions required for the rhizomes to produce quality basketry material. *Muhlenbergia rigens* benefited from burning and manual clipping as compared to the control for inflorescence production, showing that forms of management are necessary for the plants to produce abundant material for basket weavers. Furthermore, our grassland restoration project yielded interesting results as our target species germinated in the first year and reseeded in the second year. Though abundance of each species varied across treatments and years, our treatments were successful at reducing the amount of non-native grasses and increasing the percent coverage of our target species of native annual forbs. In particular, spraying herbicide two times, once in spring and once in fall prior to seeding, and spraying a single application of herbicide before a scraping treatment yielded the most promising results in restoring native annual forbs in a degraded grassland.

Introduction

For thousands of years, the first peoples of California managed the landscape. Practices such as setting fires and tending desirable plants promoted a patchwork mosaic of habitats and resources, providing the basis for California Indian material culture (Lightfoot & Parish 2009). When European settlers arrived in the late 18th century indigenous management practices were disrupted and California's native plants and people were changed forever. Today the descendants of California's first peoples are working to restore traditional knowledge, in changing ecosystems, and Pinnacles National Park is a venue for this process. Incorporating traditional management practices into degraded ecosystems has the potential to both restore ecosystem processes to which native plants are adapted, and restore cultural practices, which ensured the abundance and quality of traditional native plant resources. However, since European settlement, management practices and ecosystems have changed and have made the use and results of traditional management practices more complex.

Within Pinnacles National Park there are two important basketry plants prized by California Indian basket weavers*, Carex barbarae* and *Muhlenbergia rigens*. Both native plants need forms of management in order to provide quality material and abundant resources for basketry. *Carex barbarae* (white root sedge) is prized for its cultivated, long straight rhizomes, which are used in basketry by California Indians as stitching material (Peri & Patterson 1976; Mead 2014; Anderson 2005). Through cultivation practices such as thinning the number of plants, tilling, and removing other plant competition and rocks and debris in the soil, the rhizomes would grow straight and could reach lengths of up to 1-2 meters long, which lacked kinks, bumps, or bruises – undesirable characteristics for basketry (Peri & Patterson 1976). Historically, cultivated sedge beds had a density of about one plant per square foot (Anderson 2005). Today, white root sedge is difficult to find due to the elimination of gathering sites and problems related to gaining access to the sites, which still exist (Stevens 2006). The white root sedge that grows in sandy soils yields the highest quality rhizomes with tending, while sedge beds that are in dirt or clay are not utilized by basket weavers because even with tending they do not yield quality rhizomes (Peri & Patterson 1976).

Muhlenbergia rigens (deergrass) is also highly valued for its use in California Indian baskets. The flower stalks of deergrass were valued for their flexibility and length and were used as the stuffing in coiled baskets (Anderson 1996; Mead 2014). Deergrass flower stalks expand when soaked with water, as the stalks expanded within the basket; they tightened the stitching material and made the basket watertight (Anderson 2005). Traditional burning played an important role in the management of deergrass and it was burned every 2-5 years (Anderson 2005). Fire kept the perennial grassland free from encroachment of woody, shrubby plant species that would ultimately shade out the deergrass and kill it, since deergrass needs full sun or partial shade to survive (Daubenmire 1968; Anderson 1996). Additionally, fire would reduce the amount of old leaves which would allow more sunlight into the center of the plant which promotes new growth (Anderson 2005).

The cessation of TEK and traditional resource and environmental management (TREM) practices in California, coupled with changing fire regimes to longer fire seasons with more frequent, high intensity wildfires, has impacted the availability of native plant cultural resources for California Indians (Anderson 2005; Duguy et al. 2013; Voggesser et al. 2013). Traditional resource and environmental management practices of California Indians, such as strategic anthropogenic burning, promoted the growth of native plant species, sustained ecosystems and altered the structure and composition of plant communities (Anderson & Moratto 1996; Anderson 2005; Lightfoot & Parrish 2009; Cuthrell 2013). Thus, it is increasingly recognized that TEK and the reintroduction of historical management practices can be an important component that aids in the success of ecological restoration projects (Uprety et al. 2012; Clewell & Aronson 2013).

Traditional ecological knowledge can inform and assist ecological restoration projects in several important ways, 1) identifying and reconstructing reference ecosystems, 2) selection of species and sites for restoration projects, 3) knowledge of historical management practices, 4) management of invasive species, and 5) through post-restoration monitoring (Uprety et al. 2012).

The degradation of ecosystems accentuates the need for restoration, and the incorporation of traditional ecological knowledge with ecological restoration practices can assist restoration efforts (Uprety et al. 2012; Anderson & Barbour 2003). Many landscapes, such as California's oak woodlands, grasslands, and coastal prairies, are drastically changing, often through plant succession, in the absence of traditional ecological knowledge (Cuthrell 2013; Anderson & Barbour 2003). Additionally, grasslands in California have been dramatically transformed by non-native annual species over the past 200 years, not only displacing the native plants and the animals that depend on them, but also the rich cultural landscape of food, fiber, and medicinal resources. Hence, these ecosystems are not self-sustaining and need cultural practices to sustain them. Traditional resource and environmental management practices, such as fire and cultivation practices, were used to slow the processes of ecological succession (Kimmerer 2000), and therefore when looking to restore cultural ecosystems it is important to understand the cultural practices and processes that shaped the ecosystems that are to be restored (Storm & Shebitz 2006). For instance, Uprety (2012) and Clewell and Aronson (2013) recognize that, for thousands of years, some ecosystems coevolved with humans, whose management practices helped shape the ecosystem; hence, restoring these ecosystems should include restoration of the cultural practices that previously maintained and sustained them (Higgs 2003; Sarr & Puettmann 2000).

Ecological restoration projects would be wise to find ways to incorporate TEK into their projects, while also considering Western policies and indigenous ethics and worldviews (Uprety et al. 2012). Successful restoration of cultural ecosystems can include local and indigenous knowledge, and traditional ecological knowledge should be seen as complimentary to Western science when selecting approaches to restoration practices (Higgs 2005). When restoring cultural ecosystems, TEK can be useful for determining which plant species to plant as part of the restoration process (Anderson 2005; Uprety et al. 2012). For ecosystems that coevolved with cultural management practices, selecting species that have cultural importance may aid the success of the restoration project (Sayer 2005; Uprety et al. 2012).

California grasslands provide California Indians with many culturally important native plant species. Historically, California Indians managed grasslands through burning, digging, tending, and seed sowing (Anderson 2005). Today, the percent coverage of grasslands in California has decreased dramatically (Heady et al. 1988) and the vast majority of the remaining grasslands are degraded and dominated by non-native annual grasses and other invasive plants (Bartolome et al. 2007). Grasslands around Pinnacles National Park were probably once dominated by annual forbs (Minnich 2007; Fick & Evett 2018), which were important food resources for California Indians (Anderson 2005). Recent work to compile historical accounts (Minnich 2007) and to examine phytolith composition in older soils (Fick & Evett 2018) suggest that in the interior portions of California large swathes of historic grassland may have been dominated by annual forbs with little cover of perennial grasses. Today however, grasslands are dominated by non-native annual grasses, creating an ecosystem that did not exist prior to European contact. These annual grasslands are widespread and may represent a new stable state (Seabloom et al. 2003b), yet questions persist about whether or not these ecosystems can be restored (Harris et al. 2006).

Numerous studies have examined grassland restoration, but most have focused on reestablishing perennial grasses and few have focused on annual forbs (Stromberg et al. 2007, Holl et al. 2014a). Non-native annual grasses tend to out-compete native forbs by germinating earlier (Coleman & Levine 2007; Abraham et al. 2008; Grman and Suding 2010). When native forbs are able to germinate earlier than non-natives, they often compete successfully for space (Levine & Rees 2004; Grman and Suding 2010). Thus, methods for restoring degraded grasslands include reducing invasive plant species cover in grasslands, mowing and seed additions (Holl et al. 2014a), transplanting seedlings (Carlsen et al. 2000), topsoil removal (Buisson et al. 2006), and use of broadleaf herbicides (Lulow 2008) and fire (Stromberg et al. 2007). Among these grassland restoration methods, the most cost-effective approach is using herbicides (Holl et al. 2014b).

Researchers have had mixed results in trying to restore degraded grasslands. Holl (et al. 2014a) had little success in establishing native perennial grasses and annual forbs in degraded coastal prairie habitat from seed utilizing a variety of methods that included manually broadcasting seeds without any additional treatments, herbicide, scraping, and mowing. Buisson et al. (2008) found that removing topsoil helped to increase the establishment and survival of native perennial species by reducing competition from non-native species and nitrogen concentrations in the soil. Lulow (2008) found that native grasses and native clovers (*Trifolium* spp.) responded differently to broadleaf herbicide applications. Broadleaf herbicide application, applied prior to the introduction of native species, did not affect native grasses, but had a negative effect on native clovers.

Seed additions of native species have tended to have small effects on species composition (Clark et al. 2007). Planting from seed has the advantage of being less expensive than using transplants or plugs and often is more cost efficient for larger restoration projects. Determining seeding rates can be complicated because species vary widely in their germination rates. Therefore, seeding rates should be determined not by weight per area seeded, but by the percent live seed number. Broadcast seeding rates should generally be higher than drill seeding (Stromberg et al. 2007). Recommendations for seeding rates range from 500 – 1200 seeds per square meter depending on species for California grasslands (Stromberg et al. 2007; Holl et al. 2014a).

Together, these projects provide an excellent opportunity to expand revegetation and restoration efforts to include both native forb species and TEK in areas of Pinnacles National Park. Lessons learned in ongoing experimental projects will allow park staff to implement management strategies with proven success in restoring ecosystem structure and function, wildlife habitat, and native landscapes rich in cultural resources. The locations of this proposed work at the east entrance of the park, where the majority of visitors enter, both magnifies the problem and provides an important opportunity for the National Park Service to demonstrate environmental leadership. These management efforts aim to restore a diverse native grassland system that the park expects to become a key visitor destination.

OBJECTIVE 1:

Background and Study Area

Extensive beds of native *Muhlenbergia rigens* (deergrass) and *Carex barbarae* (white root sedge) occupy roughly 2-hectares within McCabe Canyon, a site recently acquired by Pinnacles National Park. The sedge occurs underneath a canopy of *Quercus lobata* (Valley Oak) and *Quercus agrifolia* (Coast Live Oak) along a perennial stream. Botanically, such large stands of deergrass and white root sedge are extremely rare in California due to land use changes, fire exclusion and invasive species encroachment. Culturally, deergrass and white root have deep meanings for past and present Mutsun/Ohlone and other California Indian peoples, being highly valued for use in basketry.

In addition to being culturally significant, McCabe Canyon is also botanically significant because it maintains native grassland, oak woodland, and riparian ecosystems. Valley oak savannah and riparian ecosystems are plant communities vulnerable to degradation and are considered threatened habitats by the California Native Plant Society and the California Department of Fish and Game. The area's large populations of deergrass (0.8 hectares) and white root sedge (1.2 hectares) are a rare remnant of native grasslands that were once widespread.

An important cultural and ecological process was reignited in December 2011, thanks to the initial cooperative restoration project, when tribal members gathered alongside fire crews and resource management staff to burn the 0.8-hectare stand of deergrass. California Indian people traditionally burned deergrass to clear away dead material and increase the production of flower stalks, which were highly valued in basket weaving (Anderson 2005). This was the first burn for cultural purposes conducted at Pinnacles National Park, and it marked a milestone achievement in the Amah Mutsun Tribal Band's ongoing work to revitalize their culture and restore traditional practices.

Another accomplishment resulting from the initial cooperative restoration project was the reintroduction of traditional sedge tending at Pinnacles National Park. California Indian people manage white root sedge by thinning and digging around plants to aerate and loosen soil, allowing the sedge rhizomes, which are also used as stitching in basket weaving, to grow long and straight (Anderson 2005). This activity improves the cultural resource condition of the white root sedge and allows tribal members to learn, practice, and pass on their traditional stewardship techniques.

These experiments aim to answer several questions about the management of culturally important plant species that occur in Pinnacles National Monument. Specifically, we asked the following research questions: 1) Which technique (burning, mechanical removal, or no action) is most effective at achieving the desired future conditions for deergrass for traditional uses?; 2) Does mechanical thinning of white root sedge aid in achieving the desired future conditions for practitioners?; and 3) How does the plant community respond over multiple years to various techniques to manage deergrass and white root sedge?

METHODS:

Experimental Design (white root sedge)

Within McCabe canyon, four 5 x 5-m pairs of plots were established in 2010. Pairs are closer to each other than any other plots. Four 1 x 1-m quadrats were placed 1-m in from each corner (Figure 1) in order to reduce edge effects. Each plot was calculated using the SE corner as a starting point and marked with an aluminum cap; "WRS 1A", "WRS 1B", "WRS 2A", etc. One plot from each pair was randomly selected to be tended, while the other site was left as a control.

Measuring/sampling rhizomes in the control

Sampling rhizomes in a control plot alters the plant density and other critical factors. Due to the destructive sampling of the controls, we had two adjacent areas within the control plots to sample at the beginning and end of the sample period. Sampling from control plots was done in a 1 x 1-m area, within the sample area, the middle 1 x 1-m along the southern edge.

A minimum of two practitioners assessed and scored each rhizome 'quality' in 2010. Practitioner evaluations are based on important features of rhizomes such as straightness, color, node density, and tensile strength. Practitioners also evaluated and scored rhizome samples for usability and quality using a scoring protocol that was developed as part of this research project.

Plant community characteristics

Four 1 x 1-m quadrats were placed in each plot 1-m from each corner in 2010 (Figure 1). Within each 1 x 1-m quadrat, foliar plant cover using standard point intercept measures was sampled prior to any treatment. Twenty-five points were sampled in each quadrat. Any point that did not hit living vegetation was recorded as 'no veg'. Additionally, all plant species within each 3 x 3-m sample area were listed to determine species richness of the plot.

Experimental treatment

After baseline sampling was completed in 2010 one plot was randomly selected from each pair to be treated using traditional tending practices. Treatment, conducted in spring 2011, included thinning of the vegetative material so that the white root sedge density was approximately ten plants per m^2, and removing other plant species and rocks and debris. The control plot was left untreated. Plots were re-measured during spring 2017.

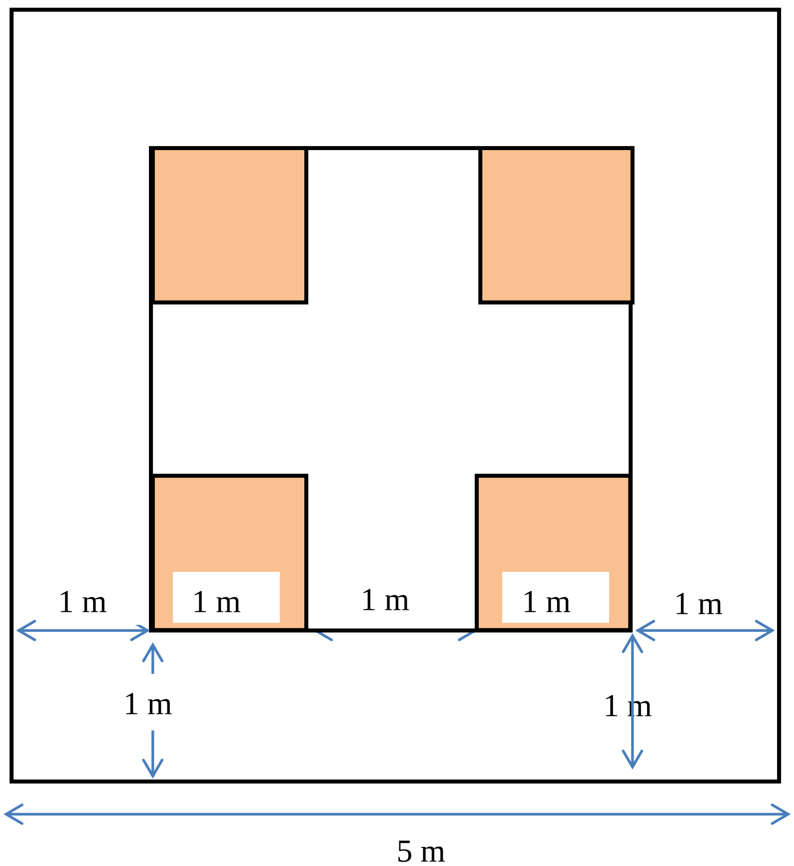


Figure 1: Plot layout for each randomly selected treatment plots. Smaller squares represent the 1 x 1-m sampling quadrats.

Experimental Design (deergrass)

Within the deergrass stand in McCabe Canyon, five blocks were established in 2010 to test the response of deergrass to burning, mechanical treatment, and no treatment. Within each block, three 2 x 2-m treatment areas were established for each treatment/control. A 2-m buffer area was established between each 2-m treatment plot and a 1-m buffer area was established outside to reduce edge effects. A 1 x 1-m sampling frame was set in the southeast corner, facing the starting point, of each plot to measure plant community characteristics and percent coverage of deergrass (Figure 2). Plots were re-measured in spring 2017.

Deergrass plant and inflorescence density

Within each 2 x 2-m treatment plot, each mature and immature deergrass plant rooted inside the plot was counted, and the number of the current year's inflorescence stalks was counted to develop the average inflorescence number per plant. For pretreatment data collection, immature deergrass were plants that showed no signs of previous inflorescence, had no thatch build up and did not have a large basal rosette like a mature plant. A mature deergrass was considered to be a plant that showed past flowering. The phenology of inflorescence stalks was noted according to the following categories:

- emergent—about to flower
- anthesis—flower emerged with protruding anthers
- fruiting—producing seed
- senescing—flowered within the past year, stalk is not totally dry and shows no signs of mold

Plant community characteristics

Within each 1 x 1-m quadrat, deergrass plant cover was sampled using standard point intercept measurements prior to any treatment. In post treatment data collection, cover of all species was measured. 25 points were sampled in each quadrat. Additionally, all plant species within the 2 x 2-m treatment plot were listed to determine species richness of the plot.

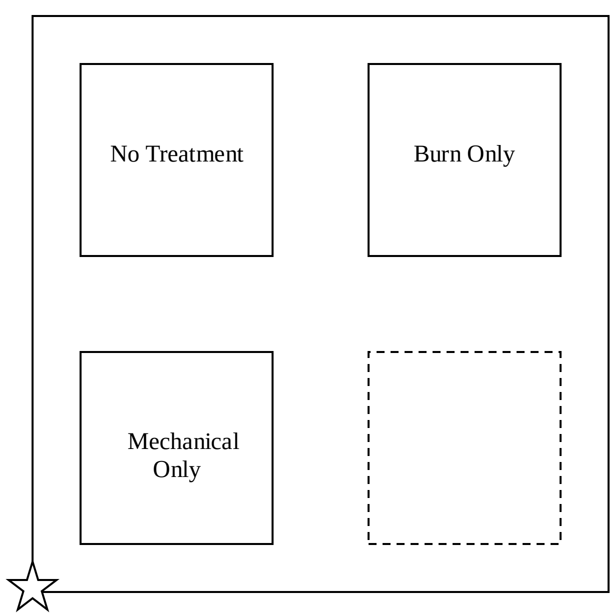


Figure 2: Block layout divided into three 2 x 2-m randomly selected treatment plots. Smaller squares represent the 1 x 1-m quadrats within each treatment area.

DATA ANALYSIS

White root sedge analysis

Total species richness and native-only species richness were analyzed for each plot each year. Data met assumptions of normality and were not transformed. Three vegetative cover variables were also selected for analysis: the cover of white root sedge in the plots, the cover of non-native species, and the cover of native species excluding white root sedge. Because white root sedge cover is a proportion with a maximum of 100% (1), and generally exceeded 70%, this variable was arcsine squareroot transformed (arcsine($\sqrt{(x)}$)). As all the plant species at each point were recorded, and the native and non-native cover data were additive, these two variables did not have an upper limit and were not proportional (in aggregate). Therefore, total cover data were not arcsine squareroot transformed. These data were skewed, so a natural logarithmic transformation was used to meet the assumption of normality.

Deergrass analysis

Deergrass inflorescence tallies, deergrass cover, and total and native species richness were chosen for analysis. Deergrass tally data was log transformed to meet assumptions of normality, and deergrass cover was arcsine square root transformed because it is proportion data. Species richness datasets were not transformed.

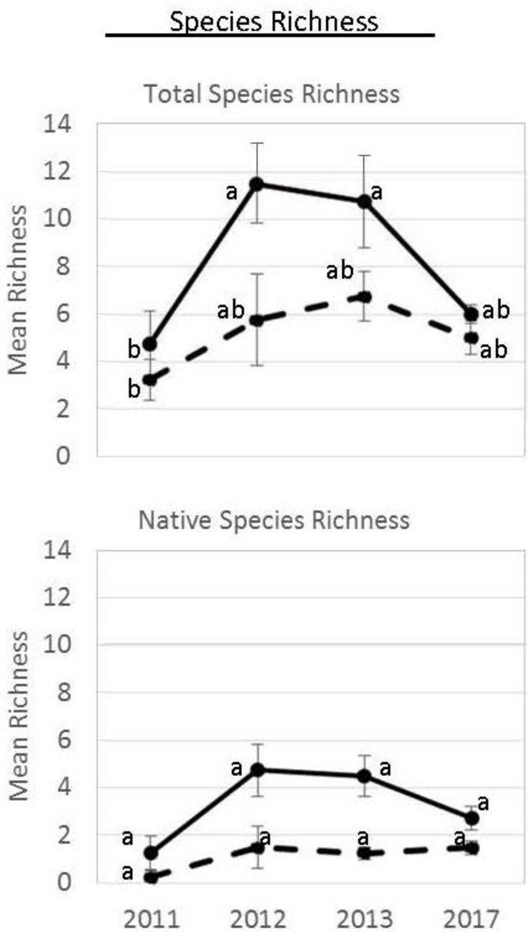
All species

A repeated measures multivariate analysis of variance (MANOVA), with measurements at each time period as the variable was originally planned. However, most data violated assumptions of sphericity, to which the repeated measures MANOVA is sensitive. Therefore, a mixed model was used instead, with year and treatment as factors and plot ID as a random effect nested within treatment. All data were run using the restricted maximum likelihood (REML) method, except white root sedge cover data. These data had negative variance estimates which did not allow REML, so the Method of Moments was used instead. *Post-hoc* tukey tests were used to distinguish differences between treatments in each year. *Post-hoc* tukey tests were used to analyze pair-wise comparisons (indicated on graphs with different letters when differences were significant).

Data were back-transformed data display. Errors for arcsine square root transformed data were determined by adding or subtracting the standard error intervals of the transformed data to the mean of the transformed data, which was then back transformed. Log natural data were also back transformed. Comparisons were considered significant if $\alpha<0.05$.

RESULTS:

White root sedge



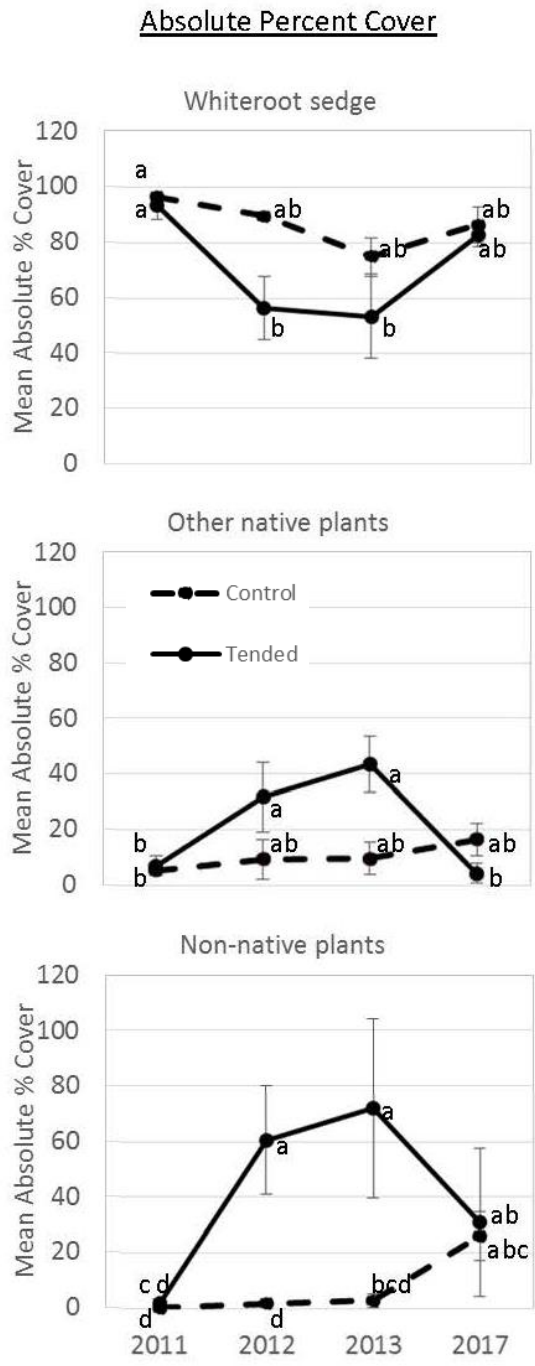


Figure 3. White root sedge plot species richness (total and native, left column) and absolute % cover (white root sedge, other natives, and non-native plants, right column) between 2011-2017 in tended (solid) and control (dashed) plots. Values for percent cover were back transformed. Mean values for each treatment are shown with error bars ±1S.E. Points with no overlap in letter are significantly different.

All five variables analyzed for white root sedge differed significantly by year (Table 1). Tended plots had significantly more native and non-native plant cover, which led to higher total species richness in the two years after treatment (2012 and 2013) relative to the year prior to treatment (2011; Figure 3). By 2017, these effects had disappeared, resulting in a significant time x treatment interaction. After tending in 2011, white root sedge cover was significantly lower in 2012 and 2013, but this difference disappeared by 2017 (Figure 3).

Table 1. Table of p-values for White-root sedge analysis variables. Significant values are in bold.

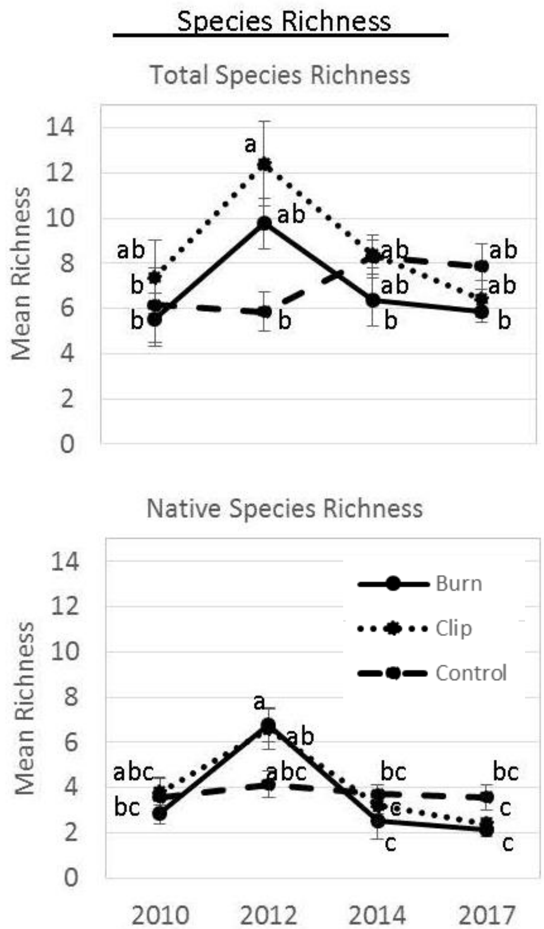
Variable	Year	Treatment	Year*Treatment
Total species richness	**0.0014**	**0.0241**	0.1897
Native species richness	**0.0066**	0.2444	0.4144
White root sedge cover	**0.0017**	**0.0112**	0.2559
Other native cover	**0.0025**	0.4680	**0.0010**
Non-native cover	**<0.0001**	**0.0003**	**0.0088**

Deergrass

Year and Treatment had significant interaction effects for all variables measured (Table 2). Deergrass cover decreased in all treatments over time, but immediately following treatment in early 2012, it was significantly lower only in burned plots (Figure 4). After the initial burning, cover in the burned plots stayed consistently at ~40%, whereas the other treatments slowly dropped in cover so that all treatment had roughly 40-50% cover in 2017, and there were no differences between treatments. Native species richness increased significantly in burned plots in the year after treatment, but this difference disappeared within one year (Figure 4). There was a trend toward the highest average inflorescence output after treatment in burned plots, but the difference was not significant (Figure 4), and changes in the average inflorescence output in the year after treatment (2012) were dwarfed by the huge increase seen in 2017 in all plots.

Table 2. Table of p-values for deergrass analysis variables. Significant values are in bold.

Variable	Year	Treatment	Year*Treatment
Total species richness	**0.0045**	0.2310	**0.0103**
Native species richness	**<0.0001**	0.7986	**0.0023**
Deergrass cover	**<0.0001**	**0.0034**	**0.0026**
Deergrass inflorescence	**<0.0001**	0.0516	**0.0789**



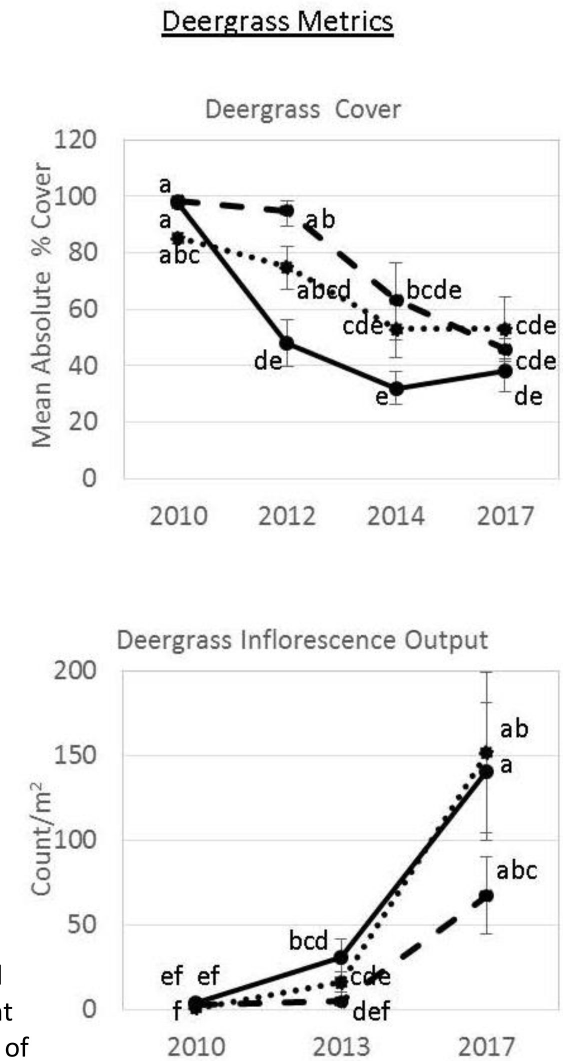


Figure 4. Deergrass plot species richness (total and native, left column) and deergrass absolute percent cover inflorescence output (right column) for each of three treatments (burn, solid line; clipped, dotted line; and control, dashed line). Values for percent cover and inflorescence output were back transformed. Mean values for each treatment are shown with error bars ±1S.E. Points with no overlap in letter are significantly different.

DISCUSSION

As exemplified by this research of deergrass and white-root sedge, TEK informed the restoration project through working with Indigenous practitioners who identified the plant species to be restored and knew the historical management practices of both plant species that promoted the abundance and quality of the desired plant parts. Julie Tex, and her two daughters, Mandy Marine, and Carly Tex from the Dunlap Band of Mono Indians continue the basket weaving practices of their ancestors and worked closely with the project participants when the project first began in 2009 to explain and demonstrate the management practices of target plant species and what constitutes usable material for baskets.

This research brought together a unique team of botanists, ecologists, horticulturalists and tribal people to further the understanding of the relationship between TREM practices and the cultivation and management of native plants so that they produce quality and abundant material for Native cultural purposes. This research also aided the Amah Mutsun Tribal Band's efforts of cultural revitalization through the relearning of TEK and TREM in collaboration with western scientists and surrounding California Indian tribes.

Our experimental treatments in the sedge beds did not yield the desired long, straight rhizomes prized by basket weavers. There are a few reasons which likely account for the lack of usable rhizomes. First, the period after the white root sedge was initially tended; California experienced its driest years in history from late 2011 to 2014. The lack of adequate water, could have affected the growth of the rhizomes. Second, tribal practitioners were not involved in the selection of the restoration sites, but rather they were randomly selected from among all sites where white root sedge was found, following standard western science practices. Upon consultation with the tribal practitioners, they would have selected sites for tending closer to the stream where the soil is sandier whereas our research sites were located away from the perennial stream where the soils had more silt and clay.

Total species and native species richness likely increased in the control plots due to the fact that prior to tending the sites, the research plots had nearly 100% cover of white root sedge. Through cultivation and thinning of the white root sedge, empty spaces with bare ground were created. These spaces were then colonized by surrounding plant species, accounting for the increase in species richness. The decrease in white root sedge percent cover was expected since part of the cultivation practice was to thin the number of white root sedge within the bed, which allows room for the rhizomes to grow straight and long.

Though the difference between the clip and burn treatments were not significant for inflorescence output of deergrass, the treatments were significant when compared to the control, particularly in 2017 which was a year of high rainfall for PNP. This suggests that forms of management are necessary for deergrass to produce an abundance of flower stalks, assuming average or above average rainfalls, and that in below average years of rainfall, deergrass does not produce abundant flower stalks. Therefore, management practices interacted strongly with interannual rainfall variation, typical of the region. There was also a significant difference between the treatments and the control for total and native species richness the year after treatments were implemented. As with the white root sedge this could be due to the presence of increased bare ground which surrounding plants colonized.

Our research shows the importance of utilizing TEK when selecting research sites that better represent Indigenous knowledge. With more strategic placement of sites based on the recommendations of the Indigenous practitioners, sites more favorable to the growth of rhizomes would have been selected. These sites may have been closer to the stream bed where the soils are sandier where rhizomes are known to grow with more vigor. We observed strong effects of thinning for a year or two after thinning but not over the longer term. This shows that white root sedge beds need ongoing maintenance, probably a minimum of every two years, to produce quality rhizomes for basket-makers. Our research also shows the importance of management to produce abundant deergrass stalks. However, it is possible with climate change and projected dry years with below average rainfall, deergrass will produce fewer flowering stalks and white root sedge will produce shorter rhizomes. California Indian basket weavers will want to harvest these native plant resources in times of average and above average rainfall to increase their chances of abundant materials.

OBJECTIVE 2:

Background and Study Area

The second experiment was conducted in the bottomlands area of the park, which is an expansive 80.9-hectare grassland system acquired by the NPS in 2005. This grassland, unlike any other area within the park, is a nearly flat, herbaceous community dominated by non-native annual species typically found in California Mediterranean Annual Grasslands. In an effort to reverse the impact from invasive species and loss of native diversity, PNP embarked on a large-scale restoration effort at the site. Since the onset of this project, the park has effectively halted the seed production and spread of the invasive yellow star-thistle (YST) throughout the area, and initiated revegetation using native perennial bunchgrasses.

For this objective, we capitalized on the site preparation of the ongoing large-scale restoration project that is taking place in the bottomlands. The site has undergone extensive land use for the past century that includes tillage, farming and cattle grazing. This intensive utilization virtually eliminated all native vascular plant taxa while disturbance dependent non-native taxa thrived. The grassland is now dominated by dense non-native annual grasses and forbs. Common non-native taxa include: *Hirschfeldia incana* (summer mustard), *Hordeum murinum* (Mediterranean barley), *Vulpia myuros* (vulpia), red-*Erodium cicutarium* (stemmed filaree), *Bromus diandrus* (ripgut brome), *Bromus hordeaceus* (soft chess), as well as the highly invasive *Centaurea solstitialis* (yellow star-thistle). Scattered native forbs are present in low numbers on the site. The most common native species on site include *Eschscholzia californica* (California poppy), *Lupinus bicolor* (annual lupine), *Amsinckia menzesii* (fiddleneck), *Distichlis spicata* (salt grass) and *Leymus triticoides* (beardless wild rye). Mature *Quercus lobata* (valley oaks) are interspersed in the grasslands.

Site preparation of the grassland was supported through the native grass restoration project. This current project refines seeding techniques used in past projects, explores some of the mechanisms to explain why certain approaches to forb revegetation are more effective than others. Our goal was is to restore patches of culturally significant wildflowers that will provide a seed source for future revegetation projects. Specifically, we proposed to answer the following research questions: 1) Which site preparation approach is most effective at establishing populations of native annual forbs? 2) What native annual species show the most promise for seeding in degraded grasslands?

METHODS:

UCSC Greenhouse Germination Tests

Pot germination tests were conducted during the winter and spring quarter to compare greenhouse and field germination numbers and to compare the germination rates of field-collected and greenhouse-grown seed. Germination tests on 11 species were started on January 13, 2017. For each species, 2012 and 2016 collections from field and greenhouse contexts were sowed in moist Promix soil, a sphagnum moss, perlite and mycorrhiza horticultural medium. Each test (n=88) used 100 seeds, divided among four replicates, resulting in 25 seed per 6.4-cm pot. The seeds were pre-cleaned and pre-counted before being placed in a 5 x 5 grid along surface of test pot. Seeds were then placed in cold stratification without light at 3.8°C and watered regularly by greenhouse staff. Germination was monitored for 11 weeks. Seeds were checked for germination every 3-4 days the first 3 weeks and every 5-7 days the following weeks. By March 15, all tests were moved from cold stratification (8 weeks).

For simplicity, a seed that exhibited any stage of germination visible with the naked eye was removed using tweezers that was sanitized in a 10% bleach solution in between tests to reduce microbial contamination. Each seedling that was removed was counted and entered into a seed germination database.

Because there were three collections of *Lasthenia* sp. and *Madia sativa*, a Tukey test was used to determine how germination differed across collections.

Greenhouse seed increase and field collection

In order to bulk up seed counts for field seeding, a seed increase for target species was performed at the UCSC Greenhouses, under the direction of Director Jim Velzy and Manager Molly Dillingham, utilizing seeds collected from previous years at Pinnacles National Park. Species grown include *Calandrinia menziesii*, *Clarkia unguiculata*, *Lasthenia* sp., *Madia sativa*, *Castilleja* sp., *Plagiobothrys* sp., *Salvia columbariae*, *Trifolium obtusifolium*, *Trifolium microcephalum*, *Trifolium wildenovii*, and *Chaenactis glabriscula*. To ensure the most abundant seed crop, proven and recommended methods of seed germination for each species were used (Emery 1988). The treatments for each species included:

- *Salvia columbariae* – soak in liquid smoke
- *Trifolium microcephalum, Trifolium obtusifolium, and Trifolium wildenovii* – soak in boiling water, then cold stratify for 2-3 weeks
- *Plagiobothrys* sp., *Lasthenia* sp., *Clarkia unguiculata, Castilleja* sp., *Calandrinia menziesii, Amsinckia menziesii,* and *Madia sativa* – cold stratify for 2-3 weeks.

Seeds were harvested at the greenhouses, and cleaned and weighed at the UCSC Arboretum, during the spring and summer of 2016. The seed increase was successful with a total of approximately 466,000 seeds produced. Additional seeds were collected on multiple days at Pinnacles National Park during spring 2016 to supplement the seed increase at the UCSC greenhouse. Amah Mutsun tribal members, UCSC Arboretum staff and students, and Pinnacles National Park staff and interns all contributed to seed collection in the field.

Experimental Design and Site Preparation

We utilized a randomized block design with five blocks (13 x 13-m) within an 8.1-hectare site within the bottomlands of PNP. Four 6 x 6-m plots were established within each block and treatments were randomly assigned. A 1-m buffer was placed between each treatment area within the blocks. We compared the efficacy of four treatments to establish native forbs in a non-native dominated California annual grassland. The four treatments included:

- a single herbicide application in fall 2016 after early germination and prior to seeding (H1)
- two herbicide sprayings (glyphosate) prior to seeding, one in spring 2016 and one in fall 2016 prior to seeding (H2).
- a soil scraping and a single fall herbicide application prior to the first rains and after scraping (S).
- a control treatment with no pre-treatment or seeding (C)

All but the control treatment was seeded with an identical seed mix as described below.

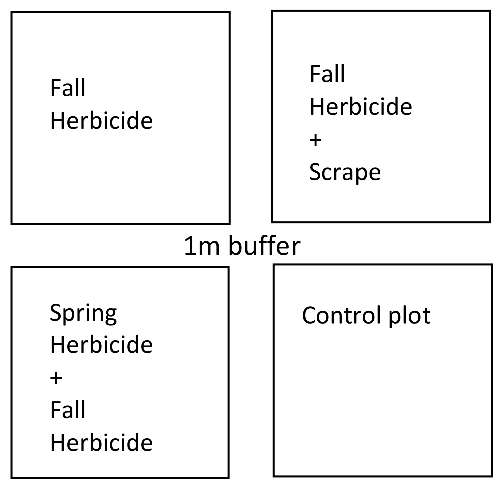


Figure 5: Plot and Sampling Design: Five replicate blocks (13 x 13-m) randomly selected within an 8.1-hectare site that was drilled in fall 2016. Plots (6 x 6-m) within each block were randomly assigned a treatment.

The spring herbicide treatment plot was sprayed during spring 2016. Scraped plot were scraped by hand with McLeods in early October. Pinnacles National Park staff sprayed all plots, except for the controls, with herbicide for the fall treatment in late November and early December 2016. All blocks were mowed and drill seeded with three native perennial bunch grasses, *Stipa pulcra* (purple needlegrass), *Poa secunda* (bluegrass), and *Melica californica* (California melic), after the fall 2016 herbicide application, between 6-8 December 2016, and a few days before a rain event.

Field Seeding

The seeds of each species were weighed and organized at the UCSC Arboretum during fall 2016 to prepare for broadcast seeding in the field. Seeds of the native annual forbs were broadcast on 12/13/2016. Seeds of all ten species were mixed in large plastic bowls with sand and then broadcast by hand into the plots. A lawn roller was used to create better contact between seed and soil.

Seeding rates of annual forbs were determined by the amount collected from greenhouse plants and collected at Pinnacles National Park during multiple years. Ultimately, to increase the possibility of success, it was determined that we would only use species with which we had enough seeds to seed at a rate of at least 50 seeds per m^2 (Table 3).

Table 3: Species used and seeding rates.

Species	Total # of seeds	Greenhouse germ rate	Seeds per m_2	Total Percent Live Seed/m_2	Total seeds per plot (36m_2)	Total seeds
Amsinkia sp.	47,604	21%	69	14.5	2,500	37500
Calandrinia menziesii	58,286	89%	85	75.7	3,066	46000
Castilleja sp.	42,121	37%	70	25.9	2,520	37800
Chaenactis glabriscula	141,073	34%	250	85	9,000	135000
Clarkia unguiculata	224,704	27%	400	108	14,400	216000
Delphinium sp.	48,470	n/a	83	41.5	3,000	45000
Eschscholzia californica	49,586	n/a	85	42.5	3,060	45900
Lasthenia sp.	251,646	53%	250	132.5	9,000	135000
Salvia columbariae	203,136	21%	213	44.7	7,692	115380
Trifolium microcephalum	120,431	17%	200	34	7,200	108000
TOTALS	1,187,057		1705	604.3	61,438	
					540m_2 total =	921580

Mowing

An additional treatment of mowing half of each plot was implemented prior to the second year of data collection as PNP staff felt this would be helpful to test the efficacy of this management method for grassland restoration. All the plots were split in half along an east-west line and randomly assigned a "mow" or "no-mow" treatment. Mowing was conducted using brush-cutters in mid-December 2017. No other treatments were performed in 2018.

Field Germination

We estimated the numbers of recently germinated seedlings for the ten seeded, native forb species on March 3, 2016 when they were mature enough to identify. We systematically placed four 0.25 × 1-m sampling frames within all plots and counted the number of individuals within the frames. If 10% or more of the area in the sample frame appeared not to have been treated with herbicide, due to errors in the application of the herbicide, the sampling frame was moved 0.5 m to the north for the southernmost plots or 0.5 m to the south for the northernmost plots.

Seedling establishment and plant community cover

On April 12, 2016, when most plants were identifiable we systematically placed four 0.25 × 1-m sampling frames in the same location as for germination estimation. Within each of the plots, we counted total number of individuals for the sown species. The density of a target weed *Hirschfeldia incana* was also measured. We also collected plant community characteristics data but with 1 x 1-m sampling frames using a standard point intercept method with 25 points. Litter and substrate were also included in the cover measurements. Additionally, all plant species within the treatment plot (excluding a 1-m buffer around the edge) were listed to determine species richness of the plot. A 5 x 5-cm sample of litter was collected at a systematic random corner outside of each sampling frame and pooled with the others in the plot, then dried and weighed to estimate the biomass of thatch on each plot and determine whether that influences plant survivorship or species composition. In 2018, we counted number of seeded native individuals and took point intercept measurements for plant guilds following the methods of 2017, but we doubled the number of quadrats to sample the mow and no-mow subplots.

DATA ANALYSIS

Data from 2017 on percent survival to adulthood of seeded species, total adult plants from seed, and total percent seed survival were analyzed using an analysis of variance (ANOVA) with treatment as a factor and *post-hoc* Tukey tests to compare the individual treatments using JMP 5.1.1 software (©SAS 2004).

The results of year two (2018) data were compared with year 1 (2017) and analyzed for both the effects of treatment and year using a mixed model with year and treatment as factors and treatment as a random effect nested within plot, also using JMP 5.1.1 software. These models were run to assess the effects of seeding using density (number of individuals per m^2) of the seeded species and two non-native species of concern, *Hirschfeldia incana* and *Centaurea sostitialis*. Plant species were pooled by functional group (native forb, non-native forb, native grass or non-native grass) and also assessed using a mixed model. Data were analyzed using the restricted maximum likelihood (REML) method, except where data had negative variance estimates, which did not allow REML. In those cases (non-native forbs cover data and the *Amsinckia*, *Calandrinia*, *Centaurea*, *Lasthenia*, *Salvia*, and *Trifolium* abundance plots), the Method of Moments was used instead. *Post-hoc* Tukey tests were used to analyze individual treatments over the two years. Differences were considered significant if $\alpha<0.05$.

We used non-metric multidimensional scaling (NMDS) to examine the relationship between the individual identities of the species forming plant community in each plot using PCOrd 4.34 (©2005 MjM Software, Gleneden Beach, Oregon, U.S.A) over the two years. Cover from the 19 most common species (present in more than 20% of plots) were analyzed and 27 uncommon species were excluded from analysis. NMDS configuration: Sorenson's distance, with random starting coordinates, 250 runs with real data, 20 runs with randomized data, stability criterion = 0.000500. 6-axes were stepped down in dimensionality to determine the best stress vs. dimensionality, and the final configuration was run with 2 dimensions. Only solutions with final stress <20 were acceptable. Multi-response Permutation Procedures (MRPP) was used to determine whether the plot locations graphed in NMDS differed significantly by treatment or year.

RESULTS:

UCSC Greenhouse Germination Tests

Four species, *Amsinckia* sp., *Chaenactis glabriscula, Clarkia unguiculata, and Lupinus bicolor* showed no difference in germination across collections (Table 4). *Calandrinia and Castilleja* had greater germination from field than greenhouse collections, whereas the reverse was true for *Lasthenia*. Older seed of *Madia* germinated less whereas other species did not show an effect of collection year. Greenhouse and field collections each produced better germination for some species, and no overall pattern was discerned.

Table 4. Germination rates in UCSC greenhouse tests. Values with different letters are significantly different across treatments.

	Seed Collections (Mean + SE)				
Species	**F12**	**F16**	**G16**	**F-value**	**Comparison**
Amsinckia sp.	19.0±1.9	21.0±0.7	~	0.3559	F12 vs F16
Calandrinia menziesii	~	10.8±3.0	2.0±0.4	**0.0280**	F16 vs G16
Castilleja exserta	10.8±0.8	~	7.8±0.8	**0.0300**	F12 vs G16
Chaenactis glabriscula	~	19.8±1.8	17.8±1.8	0.3440	F16 vs G16
Clarkia unguiculata	20.8±1.5	15.0±2.5	17.5±2.3	0.2191	Overall
Delphinium sp.	~	23.5±0.6	~	~	
Eschscholzia californica	~	6.8±1.9	~	~	
Lasthenia gracilis	6.5±0.9^{B}	9.3±0.8^{B}	14.3±1.2^{A}	**0.0009**	Overall/Tukey
Lupinus bicolor	23.0±0.4	22.8±0.5	~	0.7049	F12 vs F16
Madia sativa	14.5±1.8^{B}	21.2±0.5^{A}	20.0±1.1^{A}	**0.0099**	Overall/Tukey
Trifolium microcephalum	~	~	8.0±1.8	~	

Field Germination and Survival

All seeded species established in the first year except *Calandrinia menziesii.* All restoration treatments had higher percent survival to adulthood than the control, but there was no significant difference in seedling establishment across the two herbicide and scraped treatments in any of the target species except for *Chaenactis glabriscula*. *Chaenactis glabriscula* had significantly more survivorship in the scraped treatment than the two herbicide only treatments (Figure 6). Furthermore, herbicide and scraped treatments did not differ in terms of total number of plants from seed that survived to adulthood or percent survival of seed (Figure 7). Thatch was significantly lower in the scraped treatment, but did not differ significantly among the other treatments (Figure 8).

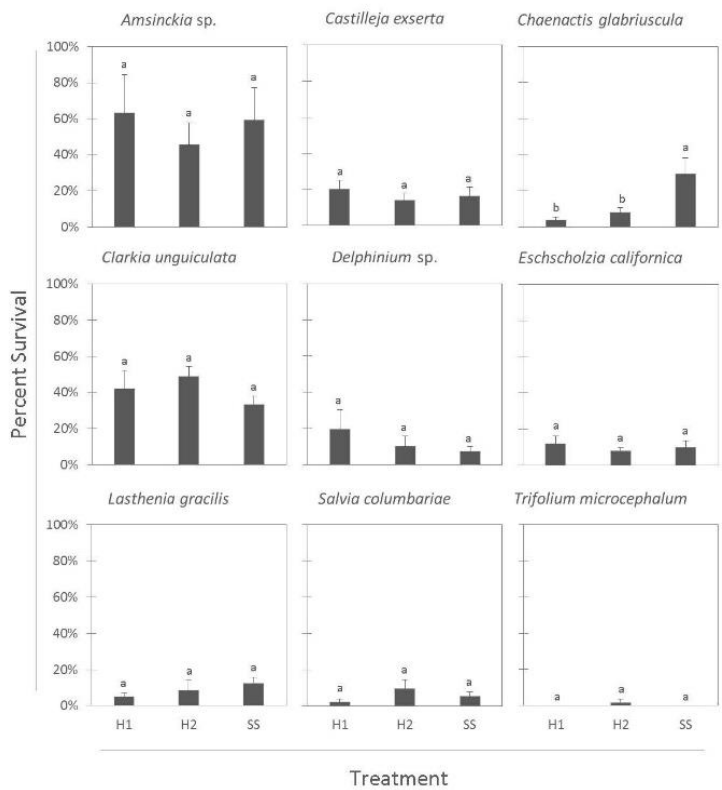


Figure 6. Percent survival to adulthood of seeded species. Percent survival = (count/m^2) / (pls/m^2) where pls (percent live seed) = greenhouse germination rate * seed/m^2. Error bars represent +1 S.E. H1 = one herbicide treatment in spring 2017. H2 = two herbicide treatments, one in spring 2017 and one in fall 2017. SS = scraping treatment

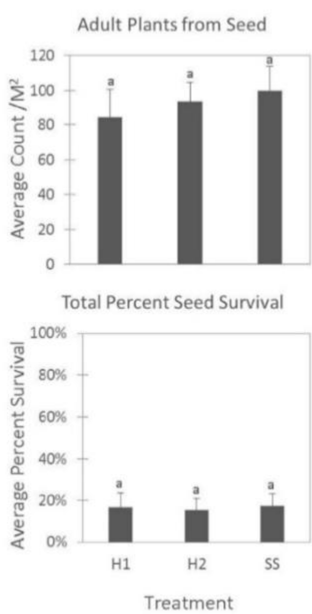


Figure 7. The top panel (Adult Plants from Seed) shows the average number of seeded plants per m^2 in each treatment. The bottom panel (Total Percent Seed Survival) shows percent survival to adulthood of total pure live seed in the plots. Percent survival = (count/m^2) / (pls/m^2). Error bars represent +1 S.E. H1 = one herbicide treatment in spring 2017. H2 = two herbicide treatments, one in spring 2017 and one in fall 2017. SS = scraped treatment

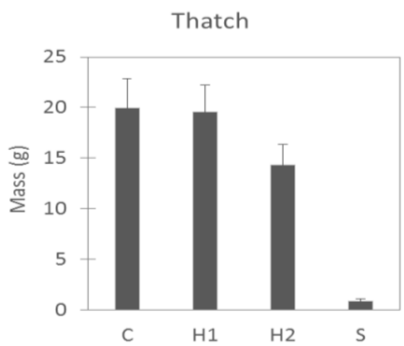


Figure 8. Mean dry biomass of thatch (g) in by treatment. Error bars represent ±1 S.E.

Density of seeded species and species of concern

All of the ten seeded species remained present in the plots in the second year after treatment. Seven of the ten seeded species and *Hirschfeldia incana* (a non-native species of concern) had significant treatment and/or year effects on seedling density. *Eschscholzia californica* had significantly higher seedling density in the three seeded plots relative to the control (Table 5). *Calandrinia menziesii* was first observed in year 2, and *Castilleja exserta*, and *Trifolium microcephalum* had significant overall increase in density in year 2 as compared to year 1 (Table 5), though individual treatments did not differ significantly (Figure 9). *Amsinckia* sp., *Clarkia unguiculata*, *Hirschfeldia incana*, and *Lasthenia gracilis* had significant interactive effects of treatment and year (Table 5) with different patterns in treatments in the two years (Figure 9). *Amsinckia* and *Lasthenia* saw significantly higher density in year 2 than year 1, as did the non-native *Hirschfeldia*, driven by significant increases in the scraped (SS) and two herbicide (H2) treatments in particular (Figure 9). *Clarkia* experienced a significant drop in cover in year 2 in the one herbicide (H1) treatment, but a significant increase in cover in year 2 in the scraped plots (Figure 9).

Table 5. P-values for the seedling density of each of the seeded species and two non-native species of concern with year and treatment as factors. Values in bold are considered significant. Non-native species have an asterisk next to their name.

	Factor p-values		
Species	**Treatment**	**Year**	**Treatment x Year**
Amsinckia sp.	0.9617	**<0.0001**	**0.0287**
Calandrinia menziesii	0.9914	**0.0344**	0.2922
Castilleja exserta	0.9247	**0.0129**	0.2469
*Centuarea solstitialis**	0.2818	0.9601	0.1011
Chaenactis glabriuscula	0.6548	0.2224	0.2932
Clarkia unguiculata	**0.0009**	0.8278	**0.0011**
Delphinium sp.	0.1408	0.1467	0.6760
Eschscholzia californica	**0.0421**	0.6353	0.1024
*Hirschfeldia incana**	**0.0059**	**0.0213**	**0.0498**
Lasthenia gracilis	0.9759	**0.0006**	**0.0010**
Salvia columbariae	0.1259	0.2672	0.6565
Trifolium microcephalum	0.9946	**0.0015**	0.1013

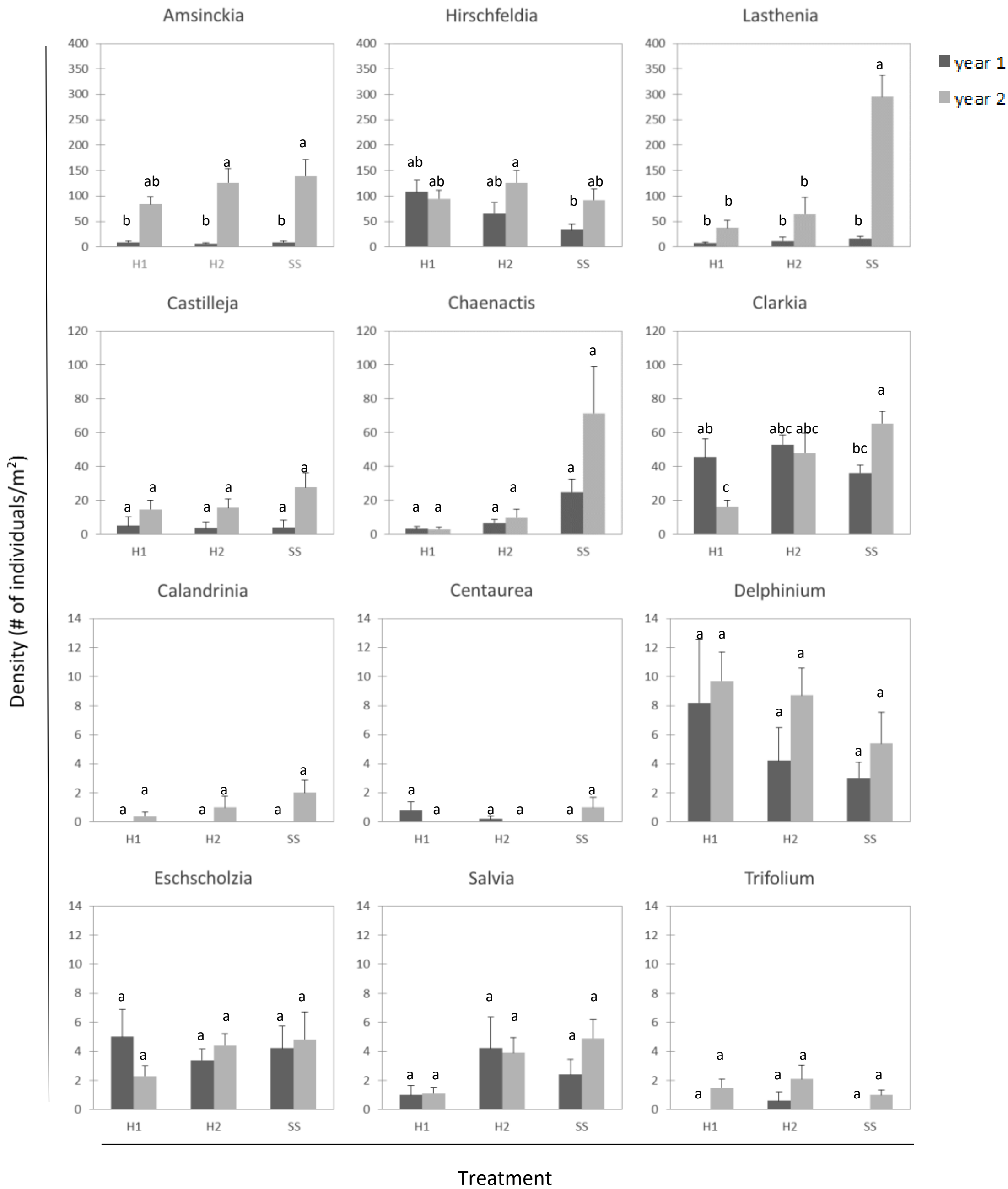


Figure 9. Density (individuals per m^2) in year 1 (2017) and year 2 (2018) of the 10 native forbs species seeded in the plot and two non-native species of concern (*Hirschfeldia incana and Centaurea solstitialis*). Mean values for each treatment are shown with error bars ±1 S.E. Points with no overlap in letter are significantly different.

<u>Plant Community Characteristics</u>

Treatment and year had interactive effects on cover of both native forbs and non-native grasses (Table 6), with native forbs decreasing significantly in year 2 in the plots with a single herbicide treatment (H1), but remaining the same in all other treatments (Figure 10). Non-native grasses decreased significantly in the control treatments (Figure 10), but showed a nonsignificant trend towards higher cover in year 2 than year 1 in the non-control treatments. Both year and treatment were significant for non-native forbs (which was primarily composed of summer mustard*, Hirshfeldia incana*). The non-control plots had higher cover of non-native forbs than the control in both years, but in year 2, average cover was higher in all the treatments than year 1. Native grass cover was very low cover in year 1 and none was measured in year 2.

Table 7 shows the species encountered in plant community monitoring. The NMDS plot (Figure 10) shows that species composition mostly clustered together by treatment and by year (final stress = 9.80, final instability =0.0012, Monte Carlo Test p=0.048. Axis 1 r^2 = 0.727, Axis 2 r^2 = 0.179). Non-/native annual grasses were more associated with the control plots, specifically the three non-native *Bromus* species, *Hordeum murinum* (foxtail barley), and *Festuca myuros* (rattail fescue). Plots were clustered more or less by treatment along axis 1 (scraped on the left, followed by Herb2, Herb1, and Control), but split by year along axis 2. *post-hoc* MRPP analysis found both treatment (p<0.0001) and year (p<0.0001) to be highly significant factors. *Chaenactis glabriscula* (CHGL) was closely associated with the scraped plots in 2017, whereas *Lasthenia gracilis* (LAGR) was associated with the scraped plots in 2018. *Eschscholzia californica* (ESCA) and *Clarkia unguiculata* (CLUN) were associated with the Herb1 and Herb2 plots in 2017, whereas in 2018 Herb 1 was associated with thatch. The density of a target weed *Hirschfeldia incana* varied by treatment, with significantly higher abundance of this species in the single-application herbicide treatment than in the scraped or control plots in 2017, but it was more associated with Herb2 plots in 2018. Non-native forbs including *Lamium amplexicaule* (LAAM), and *Hirschfeldia incana* (HIIN) were associated with Herb1 plots in 2017 and Herb2 plots in 2018 (Figure 11 and Table 7).

Table 6. P-values for cover of each functional group with year and treatment as factors. Values in bold are considered significant.

	Factor p-values		
Functional Group	**Treatment**	**Year**	**Treatment x Year**
Native forbs	**<0.0001**	**0.0312**	**0.0280**
Non-native forbs	**<0.0001**	**<0.0001**	0.2494
Native Grasses	0.0780	**0.0151**	0.3113
Non-native Grasses	**<0.0001**	0.1245	**0.0002**

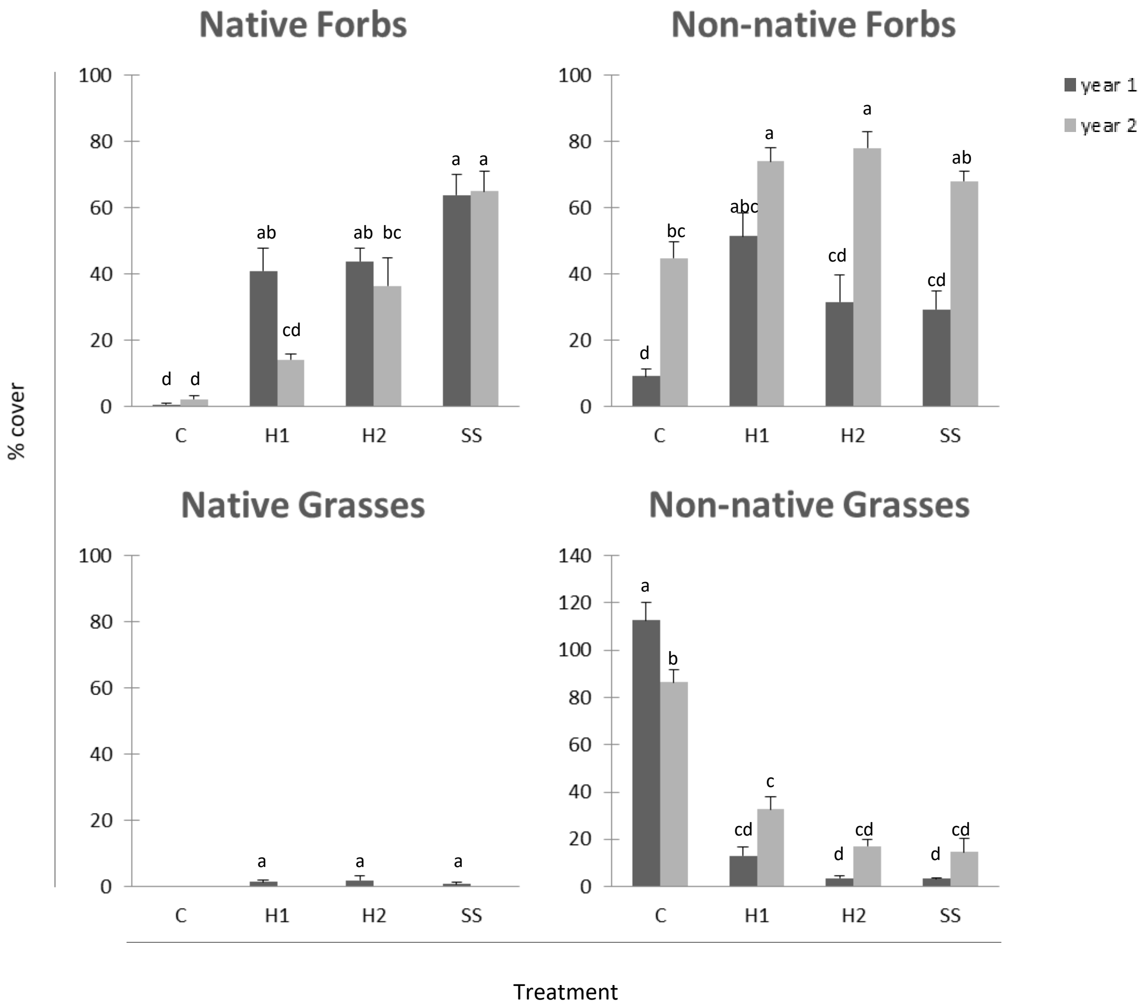


Figure 10. Percent cover in year 1 (2017) and year 2 (2018) of four major functional groups: Native forbs, non-native forbs, native grasses, and non-native grasses. Mean values for each treatment are shown with error bars ±1S.E. Points with no overlap in letter are significantly different.

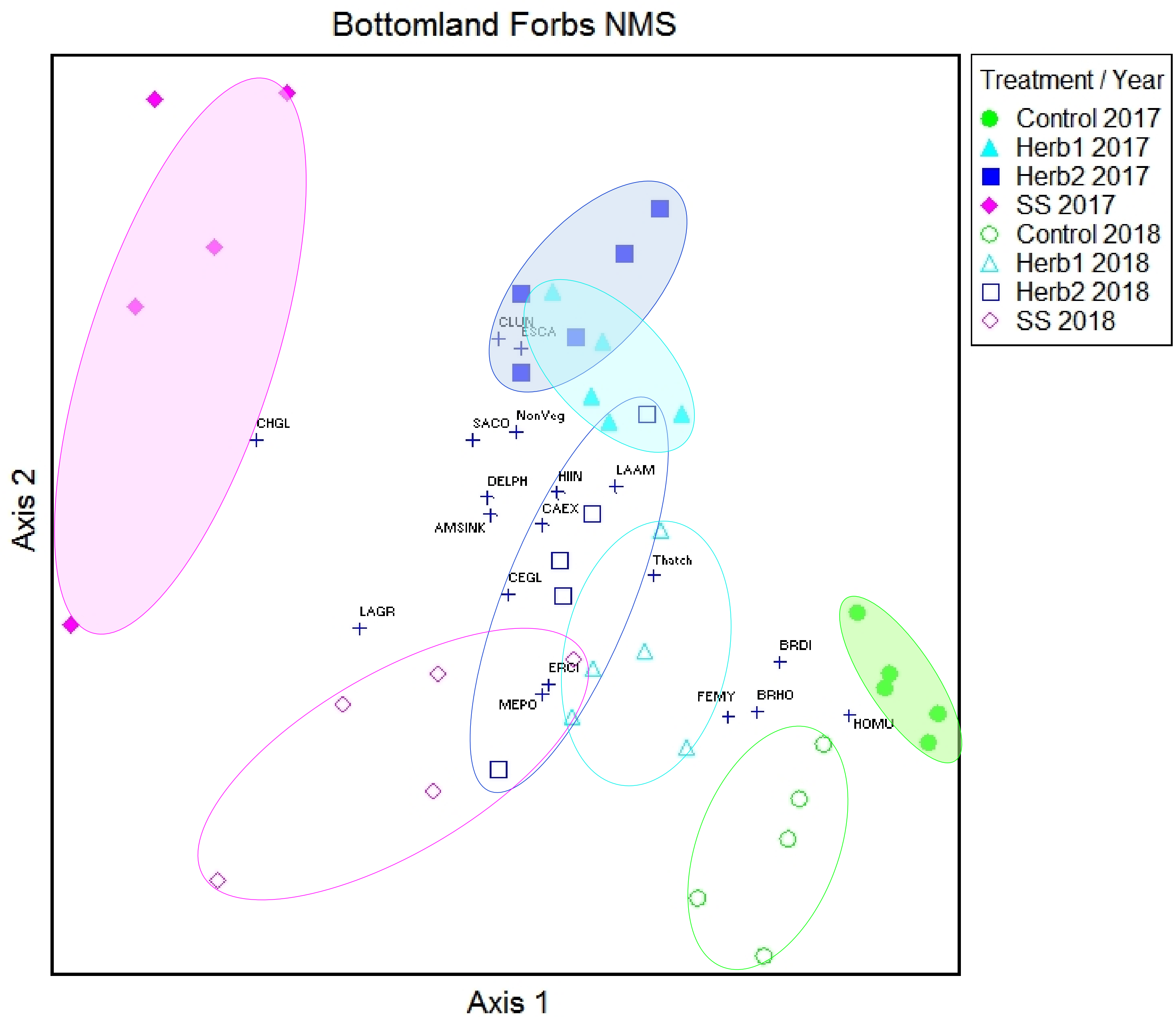


Figure 11. Non-metric multidimensional scaling graph showing the relationship of plots (circles, squares, rectangles, or diamonds) to each other based on plant community. Species are represented by crosses and their species code. Proximity to plot indicates degree of association.

Table 7. Species encountered in plant community (Figure 11) monitoring. For Duration, A = annual, and P = perennial. For Lifeform, F = forb, and G = grass. For Functional Group, NatForb = native forb, NNGrass = non-native grass, NNForb = non-native forb, and NatGrass = native grass.

Species	CODE	Native	Duration	Lifeform	Functional Group
Acmispon americanus	ACAM	Y	A	F	NatForb
Amsinckia sp.	AMSINC	Y	A	F	NatForb
Bare ground (Non-veg)	NonVeg	X	X	X	
Bromus diandrus	BRDI	N	A	G	NNGrass
Bromus hordeaceus	BRHO	N	A	G	NNGrass
Bromus madritensis	BRMA	N	A	G	NNGrass
Calandrinia menziesii	CAME	Y	A	F	NatForb
Castilleja exserta	CAEX	Y	A	F	NatForb
Centuarea solstitialis	CESO	N	A	F	NNForb
Cerastium glomeratum	CEGL	N	A	F	NNForb
Chaenactis glabriuscula	CHGL	Y	A	F	NatForb
Clarkia unguiculata	CLUN	Y	A	F	NatForb
Convolvulus arvensis	COAR	N	P	F	NNForb
Croton setigerus	CRSE	Y	A	F	NatForb
Delphinium sp.	DELPH	Y	P	F	NatForb
Erodium botrys	ERBO	N	A	F	NNForb
Erodium cicutarium	ERCI	N	A	F	NNForb
Eschscholzia californica	ESCA	Y	P	F	NatForb
Festuca myuros	FEMY	N	A	G	NNGrass
Heliotropium curassavicum	HECU	Y	A	F	NatForb
Hirschfeldia incana	HIIN	N	P	F	NNForb
Hordeum murinum	HOMU	N	A	G	NNGrass
Hypochaeris glabra	HYGL	N	A	F	NNForb
Hypochaeris radicata	HYRA	N	A	F	NNForb
Lamium amplexicaule	LAAM	N	A	F	NNForb
Lasthenia gracilis	LAGR	Y	A	F	NatForb
Logfia gallica	LOGA	N	A	F	NNForb
Lupinus bicolor	LUBI	Y	A	F	NatForb
Medicago polymorpha	MEPO	N	A	F	NNForb
Perennial grass 1	PEGR1	Y	P	G	NatGrass
Perennial grass 2	PEGR2	Y	P	G	NatGrass
Plagiobothrys canescens	PLCA	Y	A	F	NatForb
Salvia columbariae	SACO	Y	A	F	NatForb
Stellaria media	STME	N	A	F	NatForb
Thatch	Thatch	X	X	X	
Trifolium microcephalum	TRMI	Y	A	F	NatForb
Veronica persica	VEPE	N	A	F	NNForb
Vulpia myuros	VUMY	N	A	G	NNGrass

DISCUSSION

Unlike the research with *Carex barbarae* and *Muhlenbergia rigens* in McCabe Canyon where historic plant communities are largely intact and restoring traditional management practices that shaped the structure and composition of plant communities is possible, the presence of an almost entirely non-native community of plants in the bottomlands required more modern approaches to restoration. While our use of herbicides to restore native grassland species does not fit into the worldview of the Amah Mutsun Tribal Band, the use of herbicide in combination with planting native species has been proven to be effective in restoring severely degraded grasslands that have been manipulated through tillage and farming (Holl et al. 2014b). The use of herbicide applications in this project was chosen based on the efficacy of herbicide in controlling non-native grasses and forbs cost-effectively, as well as PNP's desire to use methods that they can reasonably scale up should they chose to engage in larger scale restoration efforts in the bottomlands.

Studies that have looked at restoring California grasslands have suggested that seedbank is limiting (Holl et al. 2014a; Seabloom et al. 2003a), and thus, if it is possible to succeed at restoring native forbs, enough viable seed is critical. It is in that light that our results on germination are important. Results from the greenhouse germination tests show that for the species we studied, there was no significant difference between germination rates in greenhouse grown and wild seeds collected in different years. Therefore, seeds of the selected species can likely be stored for a few years and still remain viable. However, we found that germination rates varied across species, thus it is a good practice to screen germination rates as this will help to adjust field seeding rates.

In the field all seeded species established except for *Calandrinia menziesii* in year 1, and all ten of species were found in the plots in the second year. *Calandrinia menziesii* had fairly low germination rates in the greenhouse germination tests which could be one of the reasons it wasn't observed in abundance in the field. Overall, we experienced successful establishment of most species that lasted into year 2. The fact that the first year of the project was on a relatively wet year (50cm of rainfall in 2017 compared to an average of 42cm between 1937-2016) likely had a substantial impact on the success the first year, and the buildup of sufficient seedbank to compete the second year. Because non-native annual grasses out-compete native annual forbs by germinating earlier and reducing the amount of light for native forbs (Coleman & Levine 2006; Abraham et al. 2008), the fact that it was a fairly wet year meant that in the absence of the plot treatments, native seeds likely would not have been competitive at all. Thus, due to the high amount of rainfall in the first year of the research, our success at germinating native forbs was likely due to the field conditions and "year effects" at the time in which this research was implemented (Vaughn & Young 2010), as well as the methods used prior to seeding. Furthermore, the drier conditions in the second year (24cm of rainfall were recorded in Bear Gulch in 2018), may have kept non-native cover low, which was part of the reason that the plots had such high cover of natives in the second year.

Ultimately, all ten species remained present in our treatment plots in year 2 but with varying densities. Delayed germination from the few species that were recorded in higher numbers in year 2 than year one, may have resulted from seed dormancy. The trend towards increasing non-native grasses in the plots may mean that as non-native grasses recover, they may out-compete native forbs through seed dispersal (DiVittorio et al. 2007). Non-native forbs may also pose a challenge to the continued persistence of native forbs. Treated plots had much higher cover of the non-native forbs (mainly *Hirschfeldia incana* and *Erodium* spp.) than the control in both years. The trend in the second year of increasing cover of both non-native forbs and grasses while natives hold steady, does not bode well for the long-term success of native forb populations. In these plots native forbs may be increasingly outcompeted by non-natives. However, additional years of data are needed to evaluate whether our methods established long-term populations of target species.

It is possible the presence of bare ground, created by our methods, afforded both the conditions for the native annual forbs to reseed and germinate and for non-native forbs to colonize the area. Disturbance regimes, such as mowing and cattle grazing, may increase non-native grass and forb cover (Hayes & Holl 2003a) and not shift the composition of the plant community from non-native to native species (Hayes & Holl 2003b), thus disturbance regimes alone may not be useful in restoring severely degraded grasslands (Hayes & Holl 2011). However, seeding treatments in addition to disturbance regimes may prove useful in establishing native forbs. Unfortunately, the native grasses that were drill-seeded were not detected in year 2. The reason for this is also unclear. Perhaps the native grasses did not survive into the second year due to environmental conditions.

In summary, it is clear our treatments reduced the percent cover of non-native annual grasses as compared to the control, however more data are needed to evaluate how effective our methods were long-term. Our treatments also were successful in increasing the percent cover of our selected species of annual native forbs though the percent cover and density for a given species varied. The two herbicide and scraping treatments provided the highest native cover. The presence of more bare ground in the two herbicide and scraped treatments could have facilitated the conditions favoring higher percent cover of our target species in year 2. However, this is a catch-22, because these same conditions also allowed non-native forbs to colonize the areas. Thus, a combination of disturbance regimes to control non-native annuals and forbs, both prior to and after, seeding treatments of native forbs could prove beneficial in restoring severely degraded grasslands.

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