The Parker Flats Prescribed Burn: 10th Year Post-fire Vegetation Recovery in 2015.

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Abstract

In 2000, vegetation in the 150-acre Parker Flats Parcel on Fort Ord was cleared of vegetation in order to remove unexploded ordnance. Subsequent vegetation regrowth from 2000-2005 favored plant species that are able to regenerate from below-ground buds (resprouters). There was little to no regrowth of special-status plant species that regenerate solely from seed following fire (obligate seeders). In order to improve the suitability of the Parcel as maritime chaparral habitat preserve, the Ft. Ord CRMP Group decided to burn the Parcel in 2005, with the goal of stimulating the germination and regrowth of these fire-adapted, obligate seeders from seedbank. To enhance combustion of the 5-yr old vegetation, we applied three fuel treatments prior to burning (cutting, crushing, and chaining) across twenty-four ½ acre plots (eight replicates). An additional twelve ½ acre plots were left untreated to act as controls in burned and unburned areas of Parker Flats. We have measured the cover and density of the plants at Parker Flats prior to and following the 2005 prescribed burn at 1-, 2-, 5-, and now 10-yr intervals (Pierce et al., 2004; 2005; 2006; 2007, 2010). Here we report on the 10th-year of post-fire vegetation recovery at Parker Flats in 2015.

From 2010 to 2015, the cover of native plants in maritime chaparral at Parker Flats was little changed (83% in 2010 vs. 79% in 2015). The cover of non-native species decreased from 4% to 1% as well as their distribution (59% of transects in 2015, down from 100% of transects in 2010). The cover of obligate-seeders at Parker Flats increased from <4% in 2004 to 13% in 2015, while the cover of resprouters was reduced. As expected, plant diversity (no. of species/transect) continued to decline from 2010 (31 species) to 2015 (24 species).

The 2005 prescribed fire expanded the density and distribution the HCP annual plant species at Parker Flats, although the populations of these species have declined since the 2005 burn, likely due to the increase in cover of maritime chaparral shrubs and the lack of spring rainfall in 2015. Fire also increased the density and distribution of the obligate-seeding HCP shrubs; however, the density of the resprouting HCP shrub *Ericameria facsiculata* declined in response to fire. We did not find significant differences in the cover, distribution, or density of plant species attributable to the pre-burn vegetation treatments. However, large differences in were observed between the burned and unburned plots.

Fire intensity and frequency have influenced the cover and distribution of plant species at Parker Flats. Parker Flats has its own unique fire and disturbance history - it contains areas of maritime chaparral that have been subjected to both shorter-term and longer-term fire return intervals (Pierce et al., 2007), and this appears to be reflected in distribution and density of plant species found there.

Introduction

The Parker Flats Parcel is 150 acres of maritime chaparral and oak woodland located on the former Fort Ord Army lands between Parker Flats Road and Watkins Gate Road (Pierce et al., 2004; Figure 1). In preparation for land transfer and development, the maritime chaparral in this parcel was removed in 1999-2000 using a tractor-mounted mulcher (e.g. Timberline T.A.Z.; Fecon Brush Hog), in order to clear and dispose of unexploded ordnance (UXO). Subsequently however, the parcel was re-designated as habitat preserve to mitigate for the development of lands at East Garrison (Zander Associates, 2002).

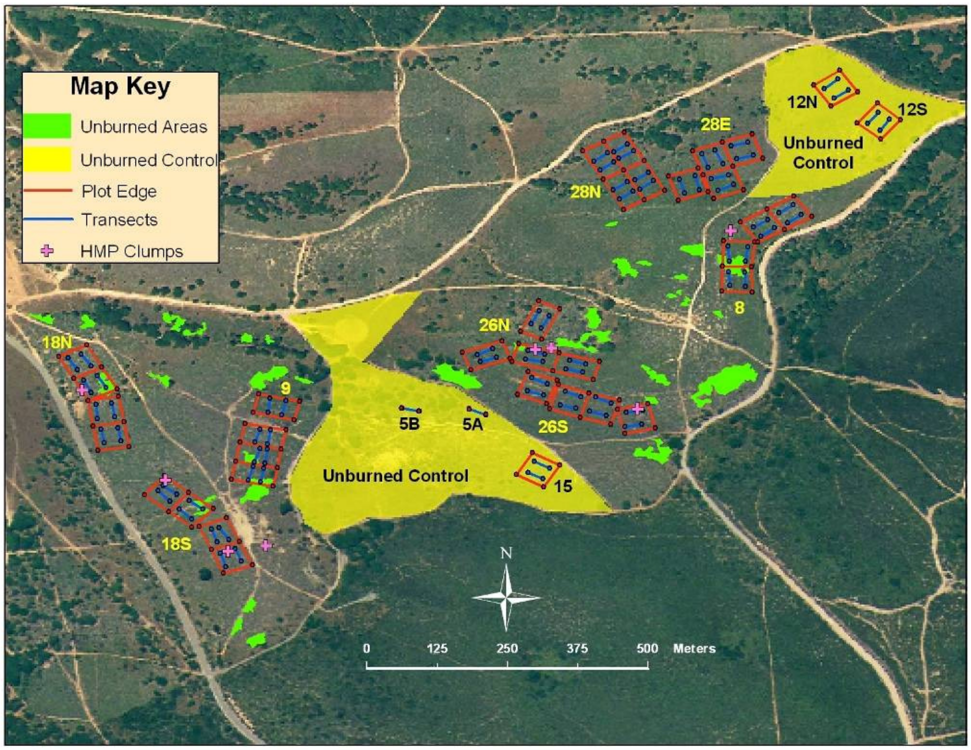

Figure 1. A map of the 150-acre Parker Flats Parcel located on the former Ft. Ord Army Lands, showing plot and transect locations in both burned and unburned maritime chaparral.

A visual survey of the density (# of plants/ac) of the special-status plant species listed in the Fort Ord Habitat Conservation Plan (HCP) was completed at Parker Flats by Jones & Stokes Associates in 1992, prior to mowing. This survey split Parker Flats into a western 46 acre polygon and an eastern 100 acre polygon. In the western polygon, medium densities (10-100 plants/ac) were reported for the HCP shrub species Monterey ceanothus (*Ceanothus cuneatus* ssp. *rigidus),* Hooker's manzanita (*Arctostaphylos hookeri),* Toro manzanita (*A. montereyensis*), and sandmat manzanita (*A. pumila*). In the eastern polygon, low densities (1-10 plants/ac) were reported for these same species, as well as for Eastwood's goldenfleece (*Ericameria fasciculata*). Monterey spineflower (*Chorizanthe pungens* ssp. *pungens*) was the only HCP annual species reported to occur at Parker Flats, with observed densities of 1-10 plants/ac in both the eastern and western polygons (Zander Associates, 2002).

In 2004, we confirmed that the density of the obligate-seeding manzanita *Arctostaphylos montereyensis* was lower in the western polygon (1.5 plants/ac) relative to the density reported in the 1992 Jones & Stokes visual survey (10-100 plants/ac), suggesting that mowing in 2000 may have reduced the density of this species (Pierce et al., 2004). In addition, we found that the maritime chaparral shrub canopy across Parker Flats was dominated by resprouting shrub species (plants that resprout from below-ground buds or from seed following fire) at the expense of obligate-seeding shrubs (plants that regenerate only from seed following fire) was low (<5%). Consequently, the Fort Ord Coordinated Resource Management Planning Group (CRMP) made the decision to burn the parcel, with the hope of re-establishing a more natural mix of plant species from the seed bank that had been established prior to mowing and UXO removal in 2000. The goal has been to use prescribed fire to release and regenerate the cover of obligate-seeders at Parker Flats to improve its utility as maritime chaparral habitat preserve.

In an attempt to dry out and enhance the flammability of the young (5-yr old) vegetation, and to create a larger range of burn temperatures that might trigger the germination of seeds of a wide-array of fire-adapted plant species, we randomly applied three pre-burn vegetation treatments (cutting, crushing, and chaining) on twenty-four ½ acre plots spread across the Parcel during the summer of 2004 (Pierce et al., 2004). In the 'cut' treatment, all plant biomass in the plot >12" above ground was cut using brush saws and laid down in place to dry. The 'crush' treatment used bulldozers to crush the plot vegetation in place. The 'chain' treatment used a 100 lb. per link chain that was dragged across the plot using two bulldozers, which acted to crush and rip up vegetation. For comparative purposes, we set up eight no-treatment control plots adjacent to the treated plots in the portions of Parker Flats to be burned. We also established four no treatment control plots in portions of maritime chaparral at Parker Flats that would remain unburned. In each plot, we established two 25m point-transects to measure plant cover, and a single 20m x 25m subplot to measure the density of the HCP shrub species (Pierce et al., 2004). We measured pre-burn plant cover and density in 2004 (Pierce et al., 2004) and 2005 (Pierce et al., 2005), burn intensity in 2005 (Pierce et al., 2005), and post-fire regrowth of the Parker Flats maritime chaparral in 2006, 2007, and 2010 (Pierce et al., 2006; 2007; 2010).

Early rains in the fall of 2004 forced us to postpone the prescribed burn until the fall of 2005. As a consequence, the treated plots to sat for a full year, allowing time for non-native annuals to invade the 8 plots that received the bulldozer crushing treatment; non-native plant cover exceeded 20% in the crushed plots relative to 1.5% in the untreated control plots. During the Fall 2005 prescribed burn, the maritime chaparral at Parker Flats burned hotter and more completely than we had anticipated. Using steel tags coated with ceramic paints that covered a range of melting points, we observed a broad distribution in burn temperatures across all treatments (200^{o}F – 1000^{o}F; Pierce et al., 2005).

In the first year following the burn (2006), we observed major increases in plant diversity and a high cover of native fire-following annuals, as well as non-native annuals, in both the treated and control plots (Pierce et al., 2006). In the second year following the burn (2007), we found that the cover of non-native annuals remained high (30-45% cover in the treated plots, 15% cover in the untreated plots), while the cover of natives had increased (Pierce et al., 2007). The cover of native species continued to increase from

2007 (55%) to 2010 (83%), at the expense of non-native annuals, which decreased in cover from 30% to 4% (Pierce et al., 2010). We found that the cover of obligate-seeding plant species had increased from 4% prior to the burn to 31% in 2010, and the no. of transects where we encountered obligate seeders increased from 19% to 77%. We have not observed significant differences in the cover or frequency of plant species between the treated plots relative to the untreated control plots. The major differences in plant cover and frequency have occurred between the burned and unburned plots.

Our objective in 2015 is to follow on from our 2010 study to continue to document how fuel treatments and fire have influenced maritime chaparral plant cover, frequency, and density 10 years after fire at Parker Flats. We compare treated and control plots, as well as burned and unburned plots, to examine how pre-burn vegetation treatments and fire have affected a) the cover, number, and frequency of plant species, and b) the density of HCP plant species at Parker Flats from 2004 to present.

Methods

In 2004, we established eight treatment blocks within the Parker Flats Parcel (Figure 1). Each block consists of four ½ acre plots randomly assigned to either the cutting, crushing, or chaining pre-burn treatment, or a no-treatment control (Pierce et al., 2004). Three of these treatment blocks were located in the 46 acres comprising western Parker Flats (burned units 9, 18N, 18S), while the remaining five treatment blocks were located in the 100 acres comprising eastern Parker Flats (burned units 8, 26N, 26S, 28E, 28N). We also set up four ½ acre plots, each with two 25m transects, in areas of maritime chaparral that were not burned (unburned units 5, 15, 12N, 12S), to actual as untreated and unburned controls. Every attempt was made to establish each of the four ½ acre plots comprising a single treatment block in homogeneous maritime chaparral. Plot corners were established using a Trimble Pro XR Global Positioning System (GPS). We established a total of 36 study plots.

Species Composition and Diversity

We mapped the end-points of our permanent plant transects using GPS in 2004. During subsequent sampling years, we re-established each of the two permanent 25m point-intercept transects across the slope in each plot based on the GPS coordinates and white PVC stakes locating the transect endpoints. The 25m transects run parallel to each other across the slope with one upslope and one downslope transect in each ½ acre study plot (Pierce et al., 2004). At each 0.5m point along each 25m transect, we recorded the dominant (first contact from top down) and secondary (additional contacts below the dominant contact) plant species in the herb (<0.5m), shrub (0.5-2m), and tree (>2m) layers, as well as the ground surface cover (bare, leaf litter, grass litter, or wood chips). Within 2.5m on either side of each transect (a 5m x 25 m area), we also recorded any additional plant species that we did not contact on the transect. We have continually utilized these same 72 transects (16 transects in each of three treated and burned plots, 16 transects in untreated and burned plots, and 8 transects in untreated and unburned plots) to document trends in species composition, frequency, and diversity between treatments from 2004 to present (Pierce et al., 2004, 2005, 2006, 2007, and 2010).

Species composition was determined for each plant layer by summing the number of point transect contacts for each species and then dividing by the total number of hits per transect (51). The number of hits for each species across each transect was then used to determine % cover by species and transect. To determine the proportion of native vs. non-native species, all recorded plant species were classified as either 'native' or 'non-native' according to the Bureau of Land Management (BLM) Ft. Ord Project Office Plant List (Styer, 2006) and the Calflora website (Calflora, 2010). We then calculated the mean percent cover by species encountered on each transect in each plot. We also calculated the frequency of occurrence for each species by summing the number of transects where that species had been contacted on the transect or noted adjacent to the transect. Transect cover samples were grouped by each treatment and control to test for treatment differences, and by burned and unburned to test for effects of fire. We compared the cover of native vs. non-native species by treatment in 2015, just as we did in previous years. We also totaled the number of species recorded both on and off the transects in each plot to document differences in species diversity between treatments. We used a one-way ANOVA (Microsoft Excel, 2003) to test for significant differences in plant cover between treatment and control plots, and between burned and unburned plots.

Density of HCP Shrub Species

As with prior HCP shrub density surveys, we used the upper 25m point transect in each ½ acre plot to establish the upslope boundary of a 25m x 20m subplot that was used count individuals of each HCP shrub species in a known area (Pierce et al, 2004). Because the HCP shrub species *Ceanothus cuneatus* ssp. *rigidus* is widespread and common at Parker Flats (>11% cover, Pierce et al. 2010), we did not count individuals of this species so that we could better focus our sampling efforts on the other four, less common HCP shrub species *Arctostaphylos hookeri*, *A. pumila* , *A. montereyensis*, and *Ericameria facsiculata*. In 2004 and 2015, we counted the number of HCP shrubs in each of these 25m x 20m subplots to obtain density estimates for each species in each study plot. Each HCP subplot is approximately 1/8 acre is size and therefore sampled 25% of the study plot. Density was then calculated for each HCP shrub species by dividing the number of individuals of each species in the subplot by the subplot area. We used this dataset to determine how fire and the pre-burn treatments have affected the density of HCP shrubs at Parker Flats from 2004 to present.

Density of HCP Annual Plant Species

As with prior surveys, there was a noticeable lack of HCP annual plant species encountered on the 25m point-transects (Pierce et al., 2006; 2007; 2010). We did however continue to find populations of sand gilia (*Gilia tenuiflora* ssp. *arenaria)* and Monterey spineflower (*Chorizanthe pungens* ssp. *pungens*) outside of our plots in the north-central portion of the Parker Flats parcel. As with prior years, we performed a broad survey of this area and flagged individual locations of each HCP annual we encountered. For each HCP annual species, we then expanded that survey out in all directions and flagged additional individuals as we encountered them. Repeating this, we were eventually able to mark the outermost boundaries of the populations of each HCP annual. We then used a Trimble handheld GPS to attain the geographic coordinates of the population boundary. We downloaded and corrected the GPS coordinates using Pathfinder Office, and plotted the population boundaries on a 2012 NAIP air photo base map using ArcGIS. For plant density estimates for sand gilia, we counted all the individuals contained in the population boundary to obtain a density estimate within the population boundary. For Monterey spineflower, we used several individual 1m2 quadrats that were randomly located within the population boundary to estimate plant density.

Results and Discussion

Species Composition and Diversity

Based on the data from the seventy-two 25m point-transects, we found that the overall plant cover at Parker Flats continued to increase from 2010 to 2015. Plant cover in the herb layer (<0.5m) averages 81% in 2015 (Figure 2), down slightly from 88% in 2010. Plant cover in the shrub layer (>0.5m) averages 57%, double the 26% cover in the shrub layer in 2010 (Pierce et al., 2010). Half of the transect points intercepted more than one species, similar to 2010. There were no significant differences in plant cover between burned treatments and control plots in either the herb or shrub layers ($p>0.05$).

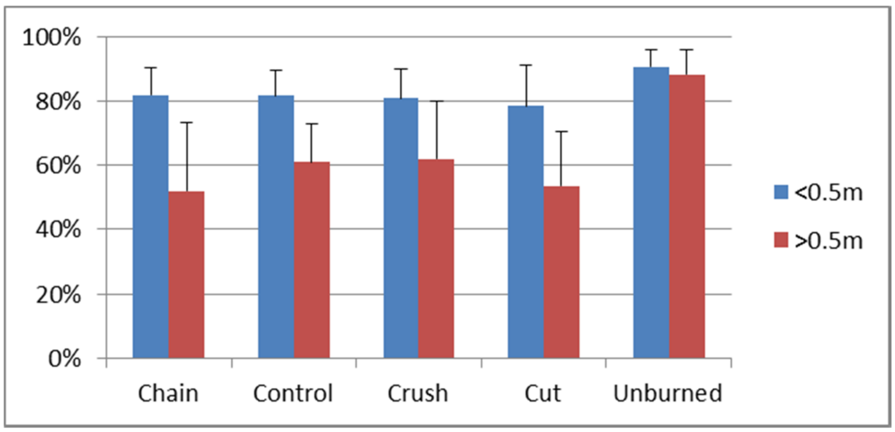

Figure 2. Percent plant cover at Parker Flats by vegetation height class and fuels treatment for 2015, 10 years after fire. Error bars represent one standard deviation from the mean.

Plant cover in the unburned plots (15 years since mowing) is significantly greater than plant cover in the burned plots ($p<0.05$; Table 1). The percent cover of bare areas (transect points with no vegetation) is higher in the burned plots (15%) relative to the unburned plots (5%), and these differences are statistically-significant ($p<0.05$). Plant cover in the <0.5m and >0.5m height classes, as well as the cover of native species, is greater in the unburned plots relative to the burned plots ($p<0.5$).

Table 1. Percent cover of plants, bare ground, by vegetation height class, and by native vs. non-native species for the treated and burned plots (Chain, Crush, Cut), the untreated and burned plots (Control), and the unburned plots for 2015. Red numbers indicate statistically-significant differences at $p < 0.5$.

% Cover by Treatment	Chain	Crush	Cut	Control	Unburned
Plants	85.7%	86.6%	82.6%	87.0%	94.6%
Bare Ground	14.3%	13.4%	17.4%	13.0%	5.4%
Plants < 0.5m in height	81.9%	80.8%	78.6%	81.6%	90.7%
Plants > 0.5m in height	52.1%	61.9%	53.7%	60.9%	88.2%
Native species (<0.5m in ht.)	79.5%	78.2%	76.8%	79.9%	90.2%
Non-native species (<0.5m in ht.)	1.7%	1.1%	0.6%	1.1%	0.0%

Native plant cover in the herb layer averages 79% (Figure 3), down from 83% in 2010. The mean cover of non-native plant species in the herb layer is 1.1%, down from 4% in 2010, and 30% in 2007 (Pierce et al., 2007, 2010). There were no statistically-significant differences in the cover of native or non-native species between the burned treated vs. untreated plots ($p>0.05$). There were also no significant differences in the cover of non-native plants in the burned vs. unburned plots ($p>0.05$). The frequency with which we contacted non-native species on the transects declined from 91% of transects in 2010 to 59% of transects in 2015. However, even if we failed to contact a non-native species on the transect, we still noted non-native plant species adjacent to all the transects (except for 1 transect in the crushed plots).

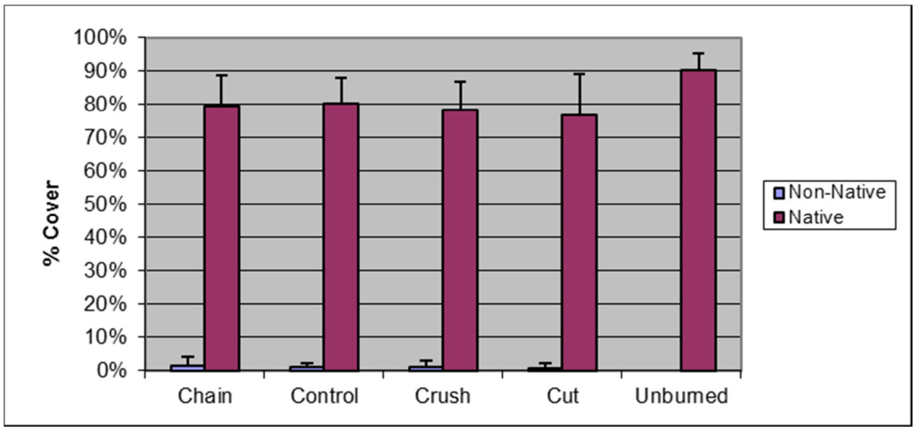

Figure 3. The cover of native vs. non-native plant species in the <0.5m height class at Parker Flats in 2015, ten years after fire. Error bars represent one standard deviation from the mean.

The mean number of species encountered on the point-intercept transects decreased from 2007 to 2015. An average of 38 plant species were encountered on the point transects in 2007, which declined to 31 species in 2010, to 24 species in 2015 (Figure 4). There were no significant differences between treatment and control plots for the mean number of species, no. of native species, or no. of non-native species encountered on the transects ($p>0.05$). Fire seems to have increased plant diversity; there were more plant species encountered on the burned transects relative to the unburned

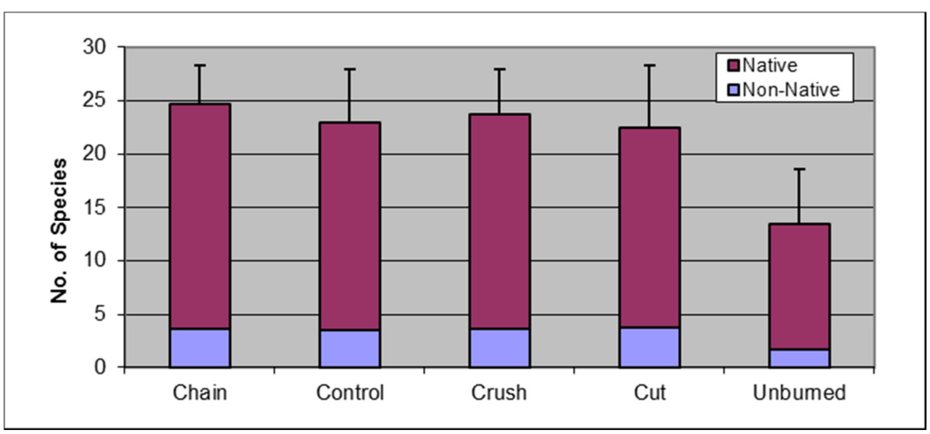

Figure 4. The mean number of plant species encountered both on and off the transects by treatment for 2015 at Parker Flats. Error bars represent one standard deviation from the mean.

transects, and these differences were statistically significant ($p<0.05$). These temporal declines in plant diversity are typical of chaparral; plant diversity in the first few years following fire is maximized and declines with time (Keeley et al., 2006). Overall, the mean number of plant species encountered per transect has fallen over time since fire, from a high of 58 species per transect in 2006 (one year following fire) to 24 plant species in 2015 (ten years following fire).

We summed the percent cover of each plant species in each height class across all plots and identified those species with a cover >0.5% (Table 2). Eleven species comprise the majority of plant cover in the burned plots at Parker Flats in 2015, all of which are native maritime chaparral shrubs. The species with the greatest % cover across all plots following fire are *Arctostaphylos tomentosa* ssp. *tomentosa* (36.9%) and *Adenostoma fasciculatum* (16.6%) in 2015. Both of these species are resprouters (ability to resprout from the root crown after disturbance). There were no statistically-significant differences in the cover of these two resprouting species between pre-burn treatments in 2015 ($p>0.05$). However, the cover of these two species in 2015 was greater in the unburned plots in comparison to the burned plots ($p<0.05$). The third most common species is the obligate-seeder *Ceanothus cuneatus* ssp. *rigidus* (9.1%). The cover of this species is greater in the cut plots relative to the chain treatment and control plots ($p<0.05$), and the cover is greater in the burned plots relative to the unburned plots ($p<0.05$). These three species were encountered on almost every transect and together comprise >60% of the plant cover across Parker Flats in 2015.

Table 2. Percent cover and frequency (% of transects) of species with an overall cover >0.5% in 2015, for the burned plots (treated + control) in 2015, unburned plots in 2015, and the untreated control plots prior to being burned in 2005 (Pierce et al., 2005).

	% Cover			% of transects		
	2015	2015	2005	2015	2015	2005
Species	burned	unburned	unburned	burned	unburned	unburned
ARTO	36.9%	50.0%	52.1%	100.0%	100.0%	100.0%
ADFA	16.6%	31.6%	17.4%	93.8%	100.0%	93.8%
CECUr	9.1%	0.7%	2.3%	96.9%	62.5%	37.5%
SAME	4.7%	3.9%	8.3%	75.0%	62.5%	62.5%
CEDE	2.0%	0.0%	0.4%	59.4%	0.0%	12.5%
BAPI	1.4%	0.2%	6.6%	68.8%	75.0%	68.8%
LECA	1.0%	0.0%	1.8%	73.4%	25.0%	31.3%
MIAU	0.9%	1.2%	2.0%	76.6%	75.0%	37.5%
CEGR	0.6%	0.0%	0.0%	25.0%	12.5%	0.0%
HESC	0.6%	0.0%	1.1%	51.6%	0.0%	25.0%
ARHO	0.5%	0.0%	0.0%	29.7%	0.0%	0.0%
Obligate Seeders	***12.7%***	***0.7%***	***3.8%***	***52.5%***	***12.5%***	***15.0%***
Resprouters	**61.6%**	**87.0%**	**88.2%**	**81.3%**	**72.9%**	**65.6%**

Species codes are ARTO = *Arctostaphylos tomentosa* ssp. *tomentosa*; ARHO = *A. hookeri*; ADFA = *Adenostoma fasciculatum*; CECUr = *Ceanothus cuneatus* ssp. *rigidus*; CEDE = *Ceanothus dentatus*; CEGR = *C. thyrsiflorus* spp. *griseus*; SAME = *Salvia mellifera*; BAPI = *Baccharis pilularis* ssp. *consanguinea*; LECA = *Lepechinia calycina*, MIAU = *Mimulus aurantiacus*, HESC = *Helianthemum scoparium*. Resprouting species are colored red, while obligate-seeding species are colored blue and *italicized*.

There have been some significant shifts in the species composition of maritime chaparral at Parker Flats since the prescribed fire in 2005. Of the eleven shrub species with a percent cover exceeding 0.5% in 2015 (Table 2), five of these species are obligate seeders (*Ceanothus cuneatus* ssp. *rigidus*, *C. dentatus, C. thyrsiflorus* spp. *griseus*, *Helianthemum scoparium*, and *Arctostaphylos hooker),* while the remaining six species are resprouters (*Arctostaphylos tomentosa* ssp. *tomentosa, Adenostoma fasciculatum*, *Salvia mellifera, Baccharis pilularis* ssp. *consanguinea, Lepechinia calycina,* and *Mimulus aurantiacus*) (Keeley and Davis, 2007; Keeley et al., 2006; personal observation, 2006). In the control plots prior to the burn in 2005 (five years after mowing in 2000), these resprouting species made up 88% of the plant cover. In contrast, the obligate seeders made up <4% of the plant cover and were encountered on only 15% of the transects (Pierce et al., 2005). Similar patterns were observed in the unburned plots in 2015 (Table 2). In contrast, the plant cover of these resprouting species is lower (62%) in the burned areas at Parker Flats in 2015, while the cover of the five obligate-seeders is substantially higher (13%, a 3-fold increase from 2005). Additionally, the five obligate-seeders were encountered across more than half of the transects in the burned plots in 2015 (a 4-fold increase from 2005; Table 2). These same patterns in cover and frequency between resprouters and obligate-seeders were observed in 2010 as well (Pierce et al., 2010).

Density of HCP Shrubs

We estimated the density of the HCP shrubs *Arctostaphylos hookeri* (ARHO), *A. pumila* (ARPU)*, A. montereyensis* (ARMO), and *Ericameria facsiculata* (ERFA) at Parker Flats by counting the number of plants of each species within a 25m x 20m subplot located in the center of each study plot. Because the HCP shrub *Ceanothus cuneatus* ssp. *rigidus* has a much greater cover (9.1%) than any of the other HCP shrub species (each <1% cover), we focused our sampling efforts on the remaining four HCP shrub species. Density was estimated as the no. of shrubs of each species in each plot divided by the subplot area (500 m2 or 0.123 acres). We counted the number of plants of each HCP shrub species in each subplot in 2004 (pre-burn; 5 years regrowth since clearance) and again in 2015 (post-burn; 10 years of regrowth since the prescribed burn). We then compare the shrub densities measured in 2004 with 2015.

We found increases in the density of all 3 *Arctostaphylos* HCP species across all treatments at Parker Flats from 2004 to 2015 (Figure 5). In contrast, the density of *Ericameria facsiculata* declined from 2004 to 2015, although this decline was not statistically significant ($p > 0.05$; $n = 32$). No single pre-burn treatment stood out as having a significant impact on density by species ($p > 0.05$; $n = 8$). The increases in density observed between 2004 and 2015 following the 2005 prescribed burn were statistically-significant across all 3 *Arctostaphylos* HCP species.

These density increases in the 3 *Arctostaphylos* HCP species relative to the density declines of *Ericameria facsiculata* can most likely be explained by differences in these species' reproductive strategies. All 3 *Arctostaphylos* HCP species are considered obligate seeders; regeneration of these species most often occurs from seed following fire, due to the effects of heat and/or charate from the fire acting to break down the seed coat (Keeley and Davis, 2007). In contrast, the wind-blown seeds of *Ericameria facsiculata* (Asteraceae) are not fire resistant; this species regenerates by resprouting from established plants as well as germinating from seed that has blown into burned sites from nearby, surviving adults (Detka, 2007). The higher fire intensities recorded during this study may be responsible for reducing the population of established plants, thereby reducing the seed source for re-establishment following fire.

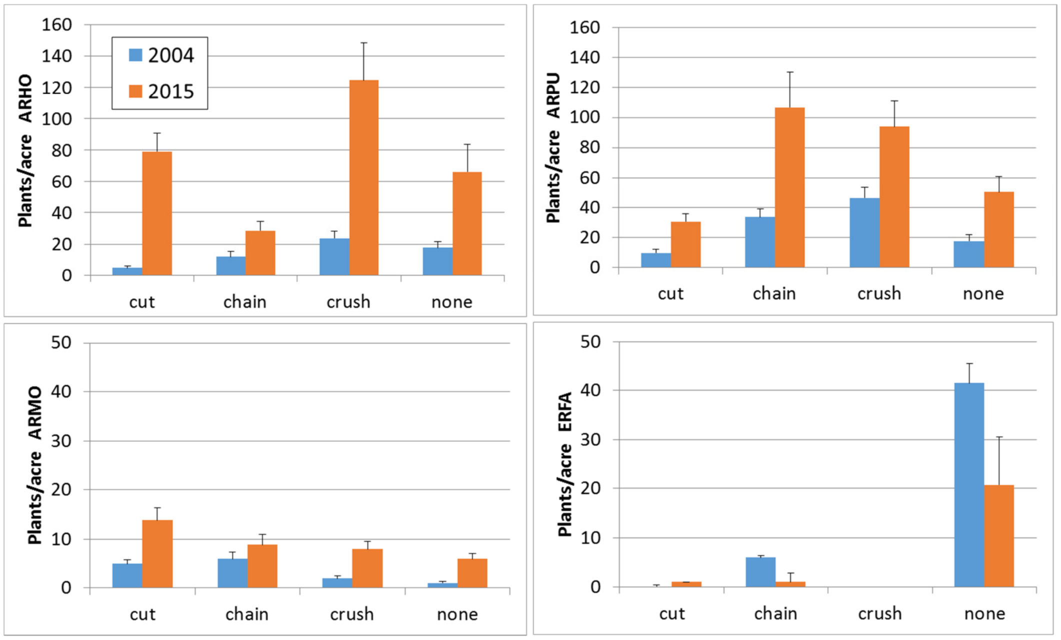

Figure 5. The density (plants/acre) of the HCP shrubs (except *Ceanothus cuneatus* ssp. *rigidus*) by treatment and the no-treatment control (none) in the 25m x 20m subplots at Parker Flats for 2004 vs. 2015. Species codes are: ARMO = *Arctostaphylos montereyensis*, ARHO = *A. hookeri*, ARPU = *A. pumila*, and ERFA = *Ericameria facsiculata.*

Fire also influenced the distribution of HCP shrub species (Table 3). The "2004 only" row of Table 3 indicates the no. of plots which lost a particular HCP shrub species after fire. In contrast, the row "2015 only" indicates the no. of plots that gained a particular HCP shrub species after fire. Fire appears to have expanded the total range of *Arctostaphylos hookeri* and *A. montereyensis* across 20% (7/32) more plots in 2015 relative to 2004, while the total range of *Arctostaphylos pumila* and *Ericameria facsiculata* remained the same. However, for all HCP species, there were a significant number of plots that gained a particular HCP shrub species from 2004 to 2015.

Table 3. Number of plots where each HCP shrub species was encountered in 2004 and 2015. "2004 & 2015" indicates the no. of plots where each species was found in both 2004 and 2015; "2004 only" indicates the no. of plots where each species was found in 2004 but not in 2015; "2015 only" indicates the no. of plots where each species was found in 2015 but not in 2004.

Year(s)	ARHO	ARPU	ARMO	ERFA
2004 & 2015	15	12	8	3
2004 only	1	4	2	2
2015 only	8	4	8	2
Change 04 to 15	7	0	6	0

Species codes are: ARMO = *Arctostaphylos montereyensis*, ARHO = *A. hookeri*, ARPU = *A. pumila*, and ERFA = *Ericameria facsiculata.*

Fire appears to have stimulated the obligate-seeding *Arctostaphylos* HCP shrubs at Parker Flats. Fire acted to increase the density of these shrubs at Parker Flats by stimulating an increase in the number of individuals encountered in any given plot. Fire also acted to increase the range of two of the four HCP shrub species we monitored by increasing the number of plots where these species were encountered. Based on this evidence, it appears that the 2005 prescribed burn at Parker Flats was successful in assisting the obligate-seeding HCP shrub species to recruit new individuals from the pre-mowing (pre-2000) seedbank.

In order to compare our data with the 1992 Jones & Stokes (J&S) visual plant surveys done at Parker Flats, we grouped the density data collected from the 25m x 20m subplots to represent the Western and Eastern Polygons outlined in the J&S survey (Table 4). Twelve study plots are located in the Western J&S Polygon ('West'; ~46 acres) and twenty study plots are located in the Eastern J&S Polygon ('East'; ~100 acres). These 25m x 20m subplots, utilized to sample the HCP shrubs and located in each study plot, sampled approximately 3.2% and 2.5% of the area in the western and eastern portions of Parker Flats, respectively.

Table 4. Density (plants/acre) of the HCP shrub species (excluding *Ceanothus cuneatus* ssp. *Rigidus*) in the Western (46 acres) and Eastern (100 acres) Jones & Stokes polygons at Parker Flats prior to brush clearing (1992), following brush clearing but prior to the prescribed burn (2004), and 10 years after the 2005 prescribed burn (2015). Species codes are: ARMO = *Arctostaphylos montereyensis*, ARHO = *A. hookeri*, ARPU = *A. pumila*, and ERFA = *Ericameria facsiculata.* The 1992 J&S surveys reported no observations of *Ericameria facsiculata* in the Western Polygon.

	West (9, 18N, 18S)			East (8, 26N, 26S, 28N, 28E)		
species	J&S	2004	2015	J&S	2004	2015
ARHO	10-100	17.5	95.8	1-10	13.4	64.8
ARPU	10-100	29.0	125.5	1-10	26.7	40.1
ARMO	10-100	4.7	4.7	1-10	2.8	12.1
ERFA	-	8.8	3.4	1-10	14.2	7.3

In the Western Polygon (units 9, 18N, 18S), the density of *Arctostaphylos hookeri* and *A. pumila* increased more than 3-fold from 2004 to 2015 following fire, while the density of *A. montereyensis* remained the same, and the density of *Ericameria facsiculata* declined. The 2015 densities of *Arctostaphylos hookeri*, *A. pumila,* and *Ericameria facsiculata* in the Western Polygon were equal to or greater than the densities reported in the 1992 J&S surveys (Zander Associates, 2002), while the 2004 and 2015 densities of *A. montereyensis* continue to be less than reported by Jones & Stokes . In Eastern Parker Flats (units 8, 26N, 26S, 28N, 28E), the density of the 3 *Arctostaphylos* HCP shrubs all increased substantially from 2004-2015 in response to fire, while the density of *Ericameria facsiculata* declined. The 2015 densities of the 3 *Arctostaphylos* HCP shrubs in eastern Parker Flats are all greater than the densities reported in the 1992 Jones & Stokes survey. In the Eastern Polygon, the densities of *Arctostaphylos hookeri* and *A. montereyensis* increased more than 3-fold, while the density of *A. pumila* doubled, and the density of *Ericameria facsiculata* declined by half.

Because the 2015 density of *Arctostaphylos montereyensis* in western Parker Flats is less than the density reported in both the 1992 Jones & Stokes survey and our own 2010 survey (Pierce et al., 2010), we were curious about the wider distribution of *A.*

montereyensis shrubs in western Parker Flats in 2015. We assembled a team of four surveyors; three to search for *A. montereyensis* shrubs and a 4th to GPS the location of each shrub found. Each of the three 'searchers' searched for shrubs of *A. montereyensis* within 2.5 meters of either side of a walking transect, which was defined prior to the start of the transect. When one of the surveyors located an *A. montereyensis* shrub, it was flagged and the GPS surveyor following behind recorded the location of that shrub. The three searchers coordinated with one another and flagged the boundaries of the survey transects to avoid sampling the same shrub more than once.

Taken together, we were able to conduct 15m wide walking transects that enabled us to survey the entire 36 acres of maritime chaparral in western Parker Flats over 3 days. We ran the survey in the Fall of 2015 to maximize the contrast between the green shrubs and the senesced annual vegetation. Prior to the walking survey, we had previously located seven *A. montereyensis* shrubs in the 25x20m HCP shrub subplots. During the walking survey, we successfully re-located six out of these seven shrubs, for a 'find rate' of 86%.

The distribution of *Arctostaphylos montereyensis* shrubs in western Parker Flats is spatially heterogeneous (Figure 6). Our walking survey found that the density of *A. montereyensis* is 20-30 plants/acre in the southern and central portions of the Western Polygon, while the northern portion has much lower densities. The densities of *A. montereyensis* appear to correlate well with the known pre-mowing locations of larger shrubs that were limbed up and allowed to stay in place, rather than being removed (Bill Collins, US Army BRAC, personal communication, 2007). The densities of *A. montereyensis* also appear to be influenced by the fire history of Parker Flats. These larger, mature individual shrubs remained alive after vegetation clearing in 2000 and were able contribute seed to the seedbank for an additional 5 years prior to the 2005 prescribed burn.

Fire frequency influences species composition of maritime chaparral (Odion and Tyler, 2002). Pierce et al., (2007) examined historical air photos from 1937 to present in an attempt to map the fire history at Parker Flats. It appears that portions of western part of Parker Flats likely experienced fires around 1970 and 1990, while the remaining portions of western Parker Flats appear to not have burned after 1950 (Figure 6). For the areas that burned in 1970 and 1990, the air photos seemed to highlight changes in the landscape which were bounded by what appeared to be fuel breaks during those two time periods. A significant portion of western Parker Flats burned around 1990, such that the vegetation in those areas was approximately 10 years old at the time the parcel was cleared in 2000. We observed little to no seed production on the 10 year-old *Arctostaphylos montereyensis* shrubs that we mapped in 2015; it's likely that *A. montereyensis* does not achieve sexual maturity until sometime after 10 years of age. Therefore, it is doubtful that the 10 year-old shrubs in the 1990 burn areas were able to achieve sexual maturity and contribute significant amounts of seed to the seed bank prior to clearing in 2000..

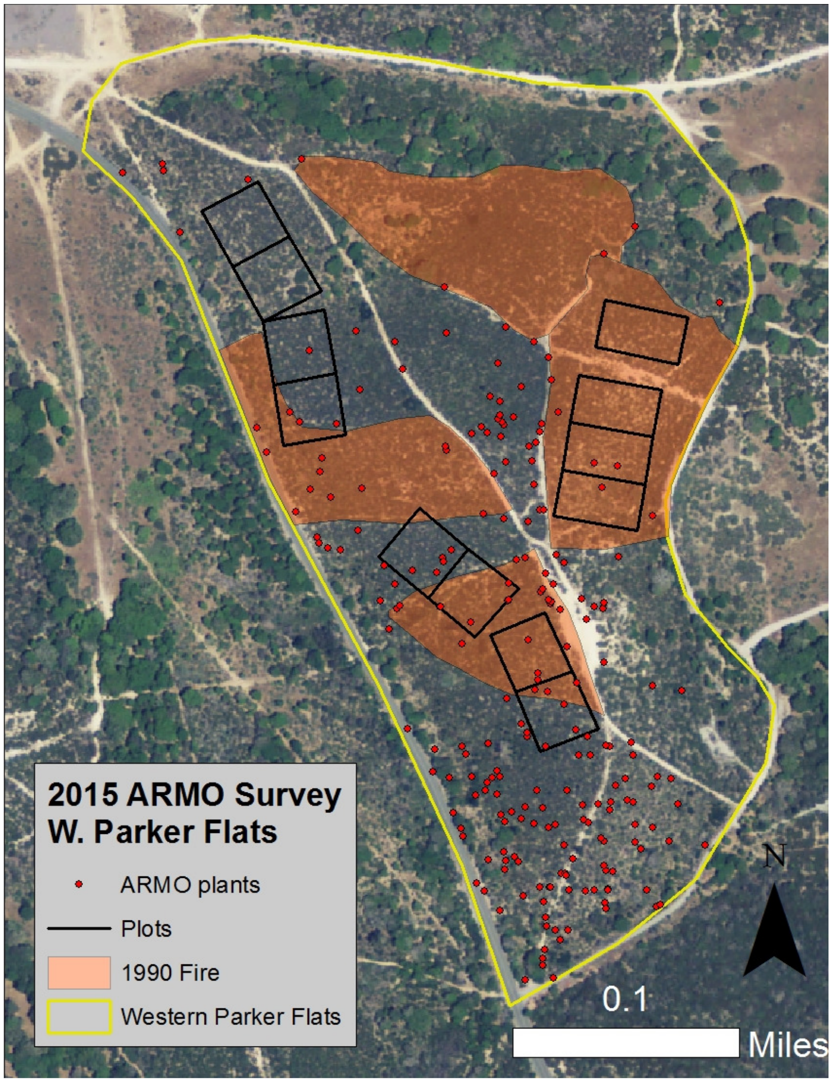

Figure 6. The location of individual shrubs of *A. montereyensis* (red dots) in western Parker Flats during 2015 relative to our study plot boundaries, and to areas that appeared to burn in 1990. The study plots outlined here are each ½ acre in size; our HCP subplots (25m x 20m), located at the center of each treatment plot, are slightly larger than 1/8th acre in size (or ~1/4 of the plot size). Note that some of the red dots contain multiple individuals.

HCP Annual Plant Species

In 2015, we again found populations of the HCP annual plant species sand gilia (*Gilia tenuiflora* ssp. *Arenaria)* and Monterey spineflower (*Chorizanthe pungens* ssp. *pungens*) within the Parker Flats Parcel (Figure 7). We did not find any individuals of coast wallflower (*Erysimum ammophilum)* like we did in prior years. The populations of Monterey spineflower and sand gilia were located in the same areas that they were found previously in the north-central portion of the Parcel (Pierce et al., 2006; 2007; 2010). This is a unique area within Parker Flats; portions appear to be grassland that is transitional to Coastal Sage Scrub, which then is transitional to maritime chaparral.

Monterey spineflower was previously reported to occur at Parker Flats prior to the burn in the Jones & Stokes 1992 survey. Sand gilia and coast wallflower had not been observed at Parker Flats until after the burn in 2006 (Pierce et al., 2006). In 2006 and 2007, we found 9 individuals of coast wallflower; that dropped to 4 individuals in 2010. In 2015, Monterey spineflower continued to be quite dense (10 plants/m2) in the grassland area that is transitional to Coastal Sage Scrub, as well as along roads and in other openings. At Parker Flats, this species seems to respond favorably to disturbance, either from fire or from the creation of fuel breaks and roads. Sand gilia has also shown a large response to fire at Parker Flats. After the prescribed burn, the population of sand gilia increased dramatically, as did the size of individual plants. As the maritime chaparral shrubs recovered in this area, both the density and the size of individuals declined from 2007 (~0.5 plants/m2) to 2015 (~0.03 plants/m2) inside of the sand gilia polygons (Figure 7).

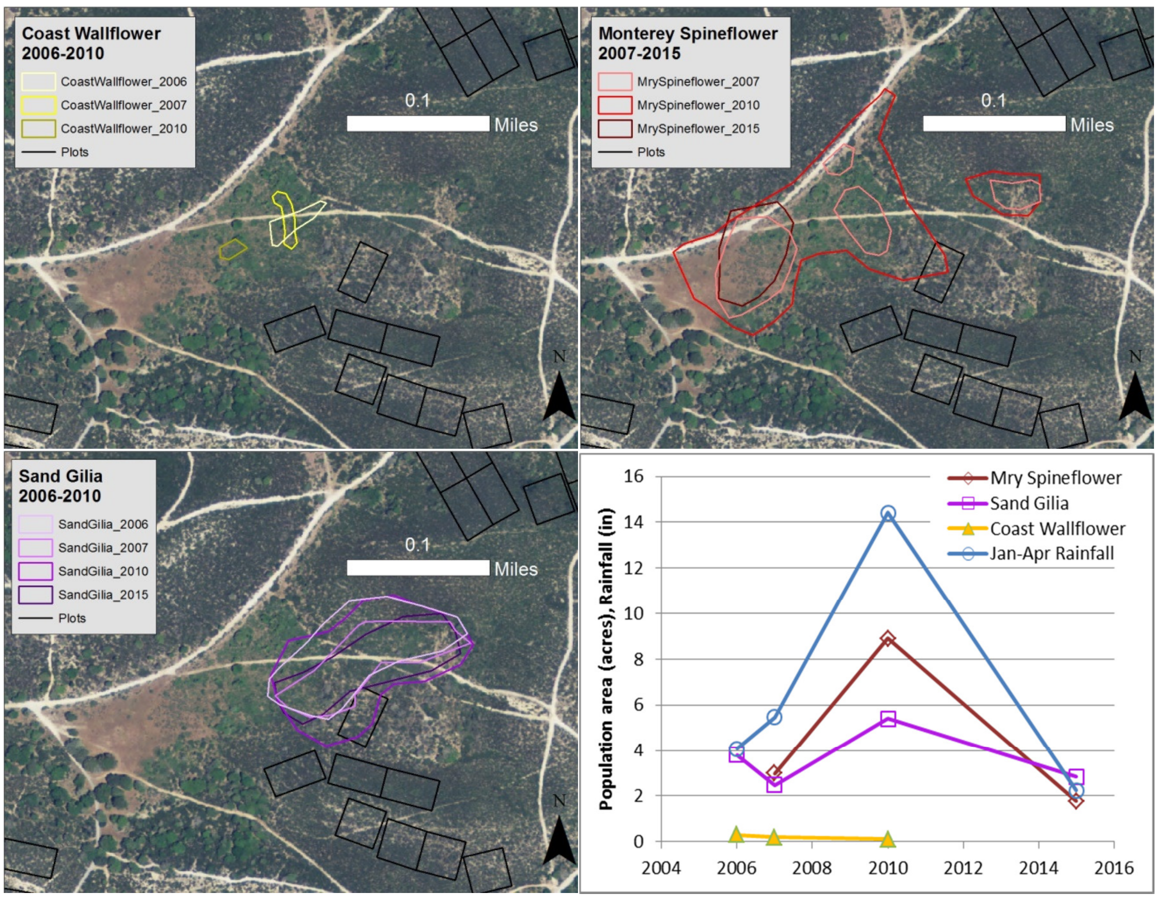

Figure 7. Locations of populations of coast wallflower (*Erysimum ammophilum)*, Monterey spineflower (*Chorizanthe pungens* ssp. *pungens*), and sand gilia (*Gilia tenuiflora* ssp. *arenaria)* within the Parker Flats Parcel from 2006 – 2015, and the relationship of population area to January-April rainfall over that same time period.

Both the populations of Monterey spineflower and sand gilia seem to respond favorably to rainfall. In years with higher January-April rainfall (e.g. 2010), the area occupied by these two species increased (Figure 7). In contrast, in years with lower rainfall (e.g. 2015), the populations of these two species (as well as other annuals) is restricted at Parker Flats. Annual variations in rainfall seem to affect the area of Monterey spineflower more than the size of individual plants.

Conclusions

Vegetation removal at Parker Flats in 2000 altered plant species composition in favor of resprouters and to the detriment of obligate seeders. The goal of this study was to use prescribed fire to regenerate these obligate seeders from the pre-mowing seedbank, and to track subsequent changes in species composition and density of the maritime chaparral at Parker Flats. The pre-burn vegetation treatments did not significantly enhance the flammability of maritime chaparral; any differences between pre-burn vegetation treatments were muted relative to the effects of burning itself on plant species composition. Prescribed fire was effective in releasing obligate seeders from the seedbank at Parker Flats. The cover, distribution, and density of obligate seeders all increased significantly following fire. The 2005 prescribed fire also enhanced the diversity of plant species found at Parker Flats, and it revealed the populations of two HCP annual plant species that were previously unknown to occur at Parker Flats.

The density of *Ericameria facsiculata* declined after the prescribed burn; this species regenerates via resprouting from existing root crowns following low intensity fires (Detka, 2007). It's unclear as to why the density of this species declined after the prescribed burn at Parker Flats – it could be because the fire intensities during the prescribed burn were high enough (200-1000 ^{o}F) to kill some of the established shrubs, which then reduced the seed source for moving into newly opened areas. The invasion of non-native grasses into areas newly opened by the prescribed burn could also have reduced soil moisture available to *E. facsiculata* seedlings. However, populations of the other HCP plant species at Parker Flats all appear to have been stimulated by fire, both in terms of distribution and density.

Fire intensity and frequency are important controls on the shrub species which comprise maritime chaparral. Higher fire intensities promote the regeneration of obligate seeding HCP shrub species, but may reduce HCP shrubs that are facultative seeders. Fire frequency relative to plant lifetimes and ages of sexual maturity can also influence the mix of plant species. Frequent disturbance (<10-15 yrs) tends to favor plant species that can achieve sexual maturity quickly and distribute seed before the next fire (e.g. *Ceanothus cuneatus* ssp. *rigidus*). Longer fire return intervals favor longer-lived species that can take several years to achieve sexual maturity but that contribute viable seed to the seedbank over many years (e.g. *Arctostaphylos montereyensis)*.

Fire history of the landscape needs to be considered when managing lands for the preservation of fire-adapted plant species. Fire frequencies should be varied across the landscape to promote the growth and cover of HCP species which vary in the length of their lifetime and time to reach sexual maturity. Ideally, a patchwork matrix of varying stand ages and fire regimes needs to be maintained across the landscape in order to manage for all of these species. As with any land parcel on Ft. Ord, Parker Flats has its own unique fire and disturbance history - it contains areas of maritime chaparral that have been subjected to both shorter-term and longer-term fire return intervals (Pierce et al., 2007), and this appears to be reflected in distribution and density of plant species found there.

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