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ORIGINAL RESEARCH

Effects of a firebreak on plants and wildlife at Pine Hill, a biodiversity hotspot, El Dorado County, California

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We examined the effects of hand clearing and pile burning of chaparral on plants and wildlife on the perimeter of Pine Hill 1–2 years after clearing. Pine Hill is home to four federally listed plant species, necessitating botanical surveys prior to clearing activities to prevent harming these species. We found over 65 new locations for the listed plants. We compared vegetation within burn scars, in cleared only plots, and in intact chaparral (control) and the germination and survival of a listed species, *Ceanothus roderickii* (Pine Hill Ceanothus). The density of *C. roderickii* seedlings was far lower in the cleared, unburned treatment than in the burned treatments, while mature *C. roderickii* was only present in unburned treatments and mature chaparral. Intact chaparral had higher cover but lower species richness of both native and exotic species than all disturbed treatments. The cleared only treatment had almost three times the cover by exotic grasses as did the burned treatments, and cover by presumably palatable, native and exotic species, was almost twice that of the burned treatments; mature chaparral contained few palatable plants, i.e. those identified as having wildlife value by Conrad (1987). Increased availability of palatable plants in treated areas may have contributed to 3–4 times higher probabilities of detection by cameras of herbivores in treated versus intact chaparral; which may have attracted smaller predators (bobcats [*Lynx rufus*] and grey foxes [*Urocyon cinereoargenteus*]) that were positively associated with western grey squirrel (*Sciurus griseus*) detections. Large carnivore detections were higher in areas with higher mule deer (*Odocoileus hemionus*) detections. Larger animals (pumas [*Puma concolor*], black bears [*Ursus americanus*], and mule deer) were detected 2–4 times more in treated areas, likely due to the physical impediment of mature chaparral. Invasion of nonnative grasses and increased wildlife usage may have been accelerated through the creation of firebreaks; continued monitoring will investigate long-term effects.

Key words: California, chaparral, firebreak, fire cued, gabbro, fuel reduction, obligate seeder, rare plants, resprouter, wildlife response to fire

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Natural fire regimes have long been altered in California (Syphard et al. 2007; Steel et al. 2015). In areas around the WUI (Wildland Urban Interface), this is even more pronounced because of the inherent danger fire represents as a management tool close to homes (Adams 2005). Since fire is frequently not an option in the WUI, fuel loads are often reduced through mastication and/or bull dozers; this disturbs soils, causes erosion (Dasmann et al. 1967; Backer et al. 2004), and provides noxious weeds a chance to become established (Giessow and Zedler 1996; Keeley 2001; Merriam et al. 2006). Additionally, plants that require a fire cue for germination may not benefit from this type of disturbance as it will not break seed dormancy (Keeley and Fotheringham 2000). Finally, the disturbance by heavy equipment may have negative effects on local wildlife populations (Bull et al. 2001; Russell et al. 2009).

The Pine Hill unit of the Pine Hill Ecological Reserve, (hereafter Pine Hill), owned and operated by the California Department of Fish and Wildlife (CDFW) in western El Dorado County, is an example of a dense, mature chaparral community in the WUI. Pine Hill is surrounded by mostly residential five-acre property parcels. To protect the surrounding private properties and Pine Hill from uncontrolled wildfire, CDFW planned to create a 20−25 m firebreak along the 4.25-km perimeter in 2017, and it was completed in 2019. This was considered necessary as uncontrolled fire may have grave ramifications on the local flora and fauna (Bennett 2003) and private property. The area is home to six special status plants that may be affected by the creation of firebreaks. Moreover, Pine Hill is landlocked and difficult to access with equipment.

The approach for creating this firebreak included measures to reduce the effects on local ecosystems that are commonly associated with firebreaks (Backer et al. 2004). We selected hand clearing and pile burning to potentially generate positive habitat effects for both plant and wildlife resources and minimize soil disturbance. In this study we examined the effects of these methods on both botanical and wildlife resources using plant surveys, vegetation plots, and trail cameras. Replicated, independent burn sites presented the opportunity to conduct research on vegetation recovery, weed invasion, and recruitment of obligately seeding plants over time.

METHODS

Study area

We conducted these studies on the perimeter of Pine Hill which is located in the center of a 120-km^2 gabbro soil island in western El Dorado County, California (Figure 1). The soil of the gabbro island is derived from the magma chamber of an ancient volcano that was pushed into the Sierra Nevada foothills by plate tectonics resulting in a soil distinctly different from the metamorphic, sediment derived soils to the west and the granitic soils of the Sierra Nevada uplift to the east (Medeiros et al. 2015). This gabbro island is home to almost 750 plant species making it one of the most botanically diverse areas on Earth (Wilson et al. 2009). Among the botanical riches are six rare species, a few of which are found nowhere else in the world; five are federally protected under the Endangered Species Act. The six rare plants are *Calystegia stebbinsii* Brummitt (Stebbins' morning-glory, hereafter CAST), *Ceanothus roderickii* W. Knight (Pine Hill Ceanothus, hereafter CERO), *Galium californicum* Hook. & Arn. ssp. *Sierrae*, Dempster & Stebbins (El Dorado bedstraw, hereafter GACAS), *Fremontodendron decumbens* R. M. Lloyd (Pine Hill Flannelbush, hereafter FRDE), *Packera layneae* (Greene) W.A. Weber & A. Love (Layne's butterweed, hereafter PALA), and *Wyethia reticulata* Greene (El Dorado mule-ears, hereafter WYRE); the first five are federally listed as threatened and/or endangered, in addition, WYRE is a federal species of concern (Table 1-1 in U. S. Fish and Wildlife Service 2002).

Pine Hill is 0.983 km^2 in size and 609 m in elevation. All rare species except CAST occur on Pine Hill mostly associated with chaparral that covers most of its aspects; the exception is GACAS which is found in or near oak woodland on the east. There is no record of when Pine Hill's perimeter last burned, and the last fire was likely many decades ago.

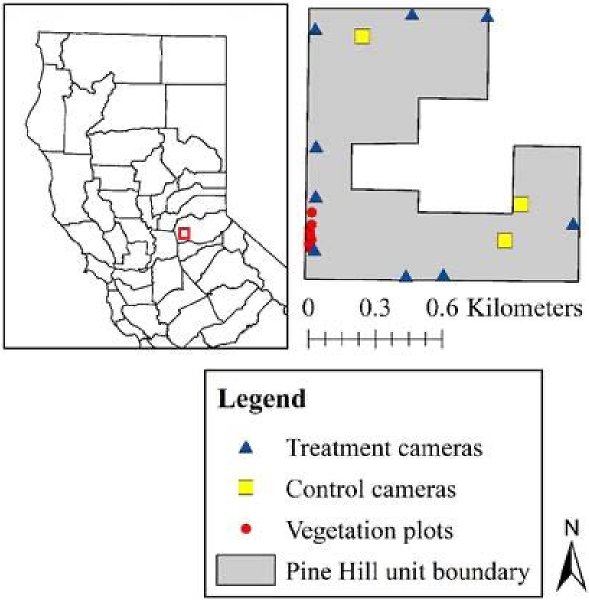

Figure 1. The locations of camera sites and vegetation plots (2017–2019) in the Pine Hill unit of the Pine Hill Ecological Reserve, El Dorado County, California are shown. The symbols used are depicted in the legend. Treatment cameras were located within burned and cleared areas in the firebreaks along the perimeter of the property, and the control cameras were located within unburned and uncleared areas of the property. The entire perimeter was surveyed for rare plants by botanists from the California Native Plant Society (CNPS). The location of the property in relation to northern California counties is outlined in the upper left.

Precipitation and seasonality are typical for a Mediterranean climate: the mean summer (June–September) temperature during the study period was 23.7°C and the mean winter (December–March) temperature was 8.8°C with average annual precipitation of 86.1 cm for 2017–2019 (NOAA 2020). On the sunnier east, south, and west slopes, chaparral shrub vegetation, dominated by white leaf manzanita (*Arctostaphylos viscida*) and chamise (*Adenostoma fasciculatum*), is found. Cooler, wetter exposures on the upper east sides and the north slopes give rise to oak woodland where black oak (*Quercus kelloggii*) and poison oak (*Toxicodendron diversilobum*) flourish.

Chaparral, a shrub-dominated plant community found in abundance throughout California, is composed of fire adapted species. These plants, including the rare species, can survive and regenerate following fire due to underground roots and stems (resprouters) or heat-tolerant seed in the soil that requires a cue to germinate by heat or chemicals released by charred wood (seeders) (Keeley and Fotheringham 2000).

No past surveys of wildlife had previously been completed at Pine Hill; however, field observations by CDFW personnel included mule deer, wild turkeys (*Meleagris gallopavo*), coyotes (*Canis latrans*), grey fox, rabbits (*Lepus* spp. and *Sylvilagus* spp.), and western grey squirrel. Further, neighboring property owner reports and observed wildlife signs suggested that the area might also be used by pumas and black bears.

Data collection

Pre- and post- clearing plant surveys.—In 2017, three 1-m wide survey paths were cut with hand tools. The paths were approximately 8 m apart while circumventing the perimeter within the footprint of the final cleared firebreak. Paths were necessary because the area did not allow for any rare plant surveying due to dense, impassable chaparral. After the trails were completed, CNPS botanists identified burn locations as well as plants of interest with a special concern for FRDE plants because of its rarity and much taller stature (>30 cm) in comparison to all the other plants. As low growing CERO was to be spared hand clearing, due to its endangered status, all plants shorter than 30 cm were left, and vegetation was cut above 30 cm. Hand clearing of chaparral and understory shrubs began in 2017 and continued to 2019; shrub piles (maximum size 10 m^2) were burned in 2017, 2018, and 2019 in locations identified as free of rare species by the CNPS botanists (see Ayres et al. 2020, this issue for survey methods). During the clearing CDFW biologists monitored Cal Fire (California Department of Forestry and Fire Protection) crews to help avoid flagged plants and to place burn piles in previously identified locations. Field assessment estimated a total of 10% of the perimeter area had experienced fire on the ground in burn piles, the remaining perimeter area was cleared only. Teams of four to six botanists surveyed the cleared perimeter for rare species in 2019-2020 and mapped new locations of CERO, FRDE, GACAS, PALA, and WYRE. The occurrence of species, and their growth stage – re-sprouting or seedling germination – within and adjacent to burned sites was noted as well.

Vegetation recovery, weed invasion, and recruitment by Ceanothus roderickii.— In May 2019, plots were established along the western perimeter in cleared areas with piles burned in 2017 (5 plots) and 2018 (3 plots); the 2017 and 2018 burn areas were next to each other and thus allowed us to compare first-year and second-year plant responses to fire within the same site. Plots sizes were the circular area of the burn pile and this area was matched in size by a cleared but unburned plot in a random direction, which was determined by the second-hand sweep of a watch, within 1 m of the burned edges (8 plots). Average plot size was 7.8 m^2; and plot sizes ranged from 4.3 to 12.65 m^2. Burned plots were selected to contain seedlings of CERO, as this rare plant is a fire obligate seeder (Boyd 2007). Rebar with a labeled PVC sleeve was placed in the center of the plots to allow ease of relocation. The diameters of the plots were measured, and plot edges were pin flagged. To assess the undisturbed chaparral vegetation, a meter tape was run into the intact chaparral parallel to the clearing and four 10 m by 10 m plots were located at 10 m intervals along the tape. An old bulldozed road/fuel break ran throughout the western aspect, approximately parallel to the western perimeter.

All species were identified, and their percent cover visually estimated for the entire plot area, as well as bare soil and rock cover for all three treatments (Barbour et al. 1998). A training session before assessments was conducted to standardize estimates among individuals. CERO seedlings were counted, and leaf number and height assessed, and a 1.5 cm long roofing nail marker was placed 1-2 cm next to each seedling. We returned in October

2019 and assessed CERO survival, i.e., seedlings next to a roofing nail, and additional germination, i.e., seedlings without a corresponding nail, and the leaf number, plant width and height of the survivors.

*Wildlife surveys.—*We sampled 12 sites (three unburned areas [controls] and nine areas within the firebreaks [treatment]) using motion-activated trail cameras (Model HC500; RECONYX Inc., Holmen, Wisconsin, USA). The cameras were placed on tree trunks and directed towards open areas; some vegetation was removed to avoid wind triggering the cameras. The camera data started on 19 March 2018 for five cameras and December 2018−February 2019 for the other 7 cameras. The data for analysis was subsampled (29 Jan 2019−11 Apr 2019, 12 Jun 2019−30 Jul 2019, 12 Sep 2019−10 Dec 2019; exceptions– camera 21: 12 Dec 2018−11 Apr 2019, 24: 11 Apr 2019−30 Jul 2019, 26: excluded 29 Jan 2019−11 Apr 2019 due to change in camera placement). The data reported here ends on 10 December 2019; however, data gathering is ongoing. We checked camera trap' batteries and replaced memory cards monthly.

Statistical analyses

All statistical analyses were completed using the program R (v.3.5.1, www.r-project.org, accessed 05 Mar 2020). To determine whether the vegetation data were distributed normally, we used the Shapiro-Wilks test (Stats v.3.6.2, www.rdocumentation.org/packages/stats/versions/3.6.2, accessed 05 Mar 2020). We also used Levene's test to determine whether the variances between the plot treatment groups were homogenous (Car v.3.0-6, www.rdocumentation.org/packages/car/versions/3.0-6, accessed 05 Mar 2020). Due to the nonparametric and heteroskedastic format of most of our data, we used the robust ANOVA tests from the R package WRS2, which used 20% trimmed means and Welch's F-statistic to reduce biases due to nonparametric and heteroskedastic data (Mair and Wilcox 2019). For all CERO seedling data (other than number of seedlings per m^2), we used two-way robust ANOVAs to analyze the difference in measurement variables between data collected on different dates and treatments, modeling treatments and dates as interaction effects. For all other vegetation measurements, we used one-way robust ANOVAs to analyze the difference in measurement variables between treatment types. For ANOVA tests that showed significant differences between groups, we used pairwise Welch's t-tests with Bonferroni's correction for multiple *P*-values to further analyze which groups were significantly different from each other (Stats v.3.6.2, www.rdocumentation.org/packages/stats/versions/3.6.2, accessed 05 Mar 2020).

Community coefficients to determine similarity/difference in plant species composition between treatments (cleared and burned compared to cleared only) and between treatments and intact chaparral (cleared and burned compared to intact chaparral; cleared only compared to intact chaparral) were calculated using the Jaccard equation (Barbour et al. 1998). This measure illustrates whether the species of two sampled areas of vegetation are of the same or different communities and can be based on either or both numbers of shared species and percent cover of shared species. Both were calculated for the plants of the treatments/intact chaparral. A Jaccard index < 50 indicates that two sampled areas of vegetation represent different communities.

Diversity was determined for each treatment (cleared and burned 2017; cleared and burned 2018; cleared only) and for intact chaparral using Shannon indices H', H'_{max}, and J', where H' is a measure of the species richness and the distribution of the species within the area sampled (e.g. across plots) combined; H'_{max} in which H' for a community is compared with the maximum possible, if all species are equally represented; and J' the ratio of H'/H'_{max}, how closely H' for the community approaches the maximum possible (Zar 1984; Legendre and Legendre 1998).

We tested for positive correlation between CERO seedling growth measurements (number of leaves, height, and diameter) using Spearman's correlation coefficient (ρ) (HMisc: Harrell Miscellaneous v.4.3-1, https://cran.r-project.org/web/packages/Hmisc/index.html, accessed 05 Mar 2020). Due to the high positive correlation ($\rho > 0.7, P < 0.05$) between seedling height and the other growth measurements, we only used seedling height as a measure of growth in our analyses.

Wildlife analyses. —To limit the biases created by having multiple detections of the same wildlife individuals, we created a second dataset (hereafter unduplicated dataset) with consecutive detections of the similar individuals (determined by species, sex, and appearance when possible) at the same camera within 30 minutes of the first detection removed (Ridout and Linkie 2009). Within the unduplicated dataset, we also removed consecutive detections of the same species within an hour of the first detection at cameras located along the firebreaks and at the two adjacent control cameras in the southern portion of the property. We used the more conservative one-hour cutoff for adjacent cameras rather than 30 minutes because individuals traveling the perimeter of the property may reach the furthest cameras in more than 30 minutes due to the distances between the firebreak cameras (0.168 km – 1.45 km). We removed these consecutive detections due to the proximity of the cameras and photo evidence of individuals being detected by multiple adjacent cameras within one hour.

To analyze whether the presence of firebreaks, other camera site location variables, and species interaction variables affected wildlife movement, we used single-species occupancy models using the methods of MacKenzie et al. (2002), and the R package unmarked (Fiske and Chandler 2011) to estimate the detection probabilities of the most abundant species (mule deer, rabbits, western grey squirrels, coyotes, grey foxes, bobcats, pumas, and black bears) at camera sites. We only measured detection probabilities because the camera sites were too close to assume spatial independence for occupancy estimates (Legendre 1993; MacKenzie et al. 2002; Lazenby and Dickman 2013). We did not document or include species' population sizes in our occupancy models because the camera sites were close together (within 1.5 km), within the same vegetation community types, and sampled at the same time (Royle and Nichols 2003). Therefore, we assumed that the population dynamics of species at each camera site were the same and would not bias the results (Caldwell and Klip 2020).

We created three groups of similar species used in the models: rabbits (*Sylvilagus* spp. and *Lepus* spp.), large carnivores (pumas and black bears), and meso carnivores (coyotes, grey foxes, and bobcats). The detections of both rabbits and large carnivores were both used as single species in the occupancy models due to the similarities between rabbit species and the low detections of large carnivore species. The detection covariates we used in the occupancy models are shown in Table 1. We calculated species detection rate covariates by dividing the number of occurrences of a species at a camera site by the number of days camera data was collected at that site. To accurately estimate the total site usage by species, we used the species detection data without duplicates removed to calculate species detection rates. When fitting models, we only included species detection rates with likely predator-prey or competition interactions with the modeled species. Prior to fitting models, we calculated Spearman's correlation coefficients for all the covariates to ensure collinearity was not an issue (Spearman's $\rho > 0.70$, $P < 0.05$). Rabbit and grey fox detection rates were correlated ($\rho = 0.74$, $P < 0.05$) and grey squirrel and grey fox detection rates were correlated ($\rho = 0.75$, $P < 0.05$). Therefor, we did not use grey squirrel and rabbit detection rates in species models that also had grey fox detection rates as a covariate.

We used the second-order Akaike's Information Criterion (AICc) (AICcmodavg: model selection and multi-model inference based on (Q)AIC(c) v. 2.2-2, cran.r-project.org/package=AICcmodavg, accessed 05 Mar 2020) to compare models and considered models with differences in AICc < 4.0 as the best approximations for the data (Burnham and Anderson 2002; Anderson 2008). We also considered models with only camera location as a covariate to further investigate control versus firebreak area usage by species. We calculated the predicted detection probabilities, regression coefficients (β), standard error, and *P*-values for the top models' covariates using the unmarked package in R (Fiske and Chandler 2011). The following equation depicts the format of the models we used to predict detection probabilities:

Table 1. Summary of detection covariates used in single-species occupancy models for mule deer, grey squirrel, rabbits, grey fox, coyote, bobcat, black bear, and puma from 12 camera sites at the Pine Hill unit of the Pine Hill Ecological Reserve, El Dorado county, California from 2018–2019.

Variable	$\bar{x}$	SE	Description
Bobcat detection rate	0.011	0.018	Number of bobcats detected per camera per day.
Coyote detection rate	0.01	0.005	Number of coyotes detected per camera per day.
Human detection rate	0.056	0.085	Number of humans detected per camera per day.
Grey fox detection rate	0.024	0.039	Number of grey foxes detected per camera per day.
Grey squirrel detection rate	0.031	0.044	Number of grey squirrels detected per camera per day.
Large carnivore detection rate	0.007	0.012	Sum of number of pumas and black bears detected per camera per day.
Meso carnivore detection rate	0.046	0.054	Sum of number of grey foxes, coyotes, and bobcats detected per camera per day.
Mule deer detection rate	0.449	0.315	Number of mule deer detected per camera per day.
Rabbit detection rate	0.042	0.049	Number of rabbits (cottontail and jackrabbits) detected per camera per day.
Camera location			Defined by location of camera in the firebreak (burned) areas or unburned areas.
Property side			Defined by location of the camera relative to the closest side of the property (north, south, east, or west side).
Ordinal day			Ordinal date of each camera day. Range of 1–365.
Season			Season of each camera day (summer, fall, winter, spring).

$$logit(p_{ij}) = \beta 0_i + \beta 1_i\,(covariate\ 1)_j + \beta 1_i\,(covariate\ 2)_j + \cdots + \beta n_i\,(covariate\ n)_j\,,$$

where *logit* is the logarithm of the odds, p_{ij} is the detection probability of species i at site j and covariates 1 through n are the values of each covariate used in the model at site j.

RESULTS

Plant surveys

As the perimeter of Pine Hill had never been comprehensively botanically surveyed, many of our findings were novel. A total of 120 rare plant locations on public land were recorded; 65 of these points were individual plants of FRDE, and patches of CERO, GACAS, PALA, and WYRE that had not previously been reported to the California Natural Diversity Database (CNDDB) or CalFlora.

Except for GACAS, all species were present or reported from chaparral communities on the sunnier aspects, and were mostly absent from the north aspect and under oak woodland; the exceptions being GACAS which was found only under the east side oak woodland canopy, three FRDE plants on the north side, and WYRE which grew under every aspect and condition (Table 2).

Table 2. Summary of rare plant species distributions on the perimeter of the Pine Hill unit of Pine Hill Ecological Reserve, El Dorado County, California from March 2017 to April 2019. All species were located with GPS and mapped except CHGR which was too numerous to map. Species are as follows: CERO (*Ceanothus roderickii*), CHGR (*Chlorogalum grandiflorum*), FRDE (*Fremontodendron decumbens*), GACAS (*Galium californicum ssp. sierrae*), PALA (*Packera layneae*), WYRE (*Wyethia reticulata*). FRDE was reported on private property on the east but was not found by the survey teams.

Species	New locations	Aspect				
		East-chaparral	South	West	North	East-oak woodland
CERO	10	Present	Present	Present	Absent	Absent
FRDE	6	Reported	Present	Present	Present	Absent
GACAS	16	Absent	Absent	Absent	Absent	Present
PALA	15	Present	Present	Present	Absent	Absent
WYRE	18	Present	Present	Present	Present	Present

Vegetation recovery, weed invasion, and recruitment by Ceanothus roderickii. —A total of 72 species were recorded across all treatments; 51 were native and 21 were exotic. All shrubs and trees were native. Jaccard community coefficient between intact chaparral and the burned and cleared treatment illustrated different communities, reflecting the low cover and number of shared species between them. Jaccard between the burned and cleared treatment and the cleared only treatment also showed different communities especially based on cover of shared species, reflecting the difference in treatment effects on establishment and growth (Table 3).

Highest Shannon diversity was found in the cleared and burned treatment - where burning occurred in 2017 - and in the cleared only treatment, reflecting the high species richness of the two treatments (39 and 32 species, respectively) and relatively even distribution of species across sampled plots. The lowest Shannon diversity was found in the intact chaparral, due to its lower species number (18), and the differences in species composition in each plot along the transect (8 of the 18 species were recorded in only 1 of the 4 plots of the transect; Table 4).

Table 3. Jaccard community coefficients of vegetation of paired treatments (burned and cleared, cleared only) and intact chaparral, calculated based on percent cover of shared species and on number of shared species (spp. #). Coefficients less than 50 indicate different communities between pairs. Data collected at the Pine Hill unit of the Pine Hill Ecological Reserve, El Dorado County, California in May and October 2019.

Community coefficients	By % cover	By spp. #
Jaccard intact chaparral with burned and cleared	5	25
Jaccard intact chaparral with cleared only	20	28
Jaccard burned and cleared with cleared only	20	46

Table 4. Diversity indices (Shannon) for each treatment (cleared and burned 2017, cleared and burned 2018, cleared only) and for intact chaparral. Higher H' and J' indicate higher species richness and evenness of species distribution across plots combined. H'_{max} indicates highest H' possible. Data collected at the Pine Hill unit of the Pine Hill Ecological Reserve, El Dorado County, California in May and October 2019.

Treatment	H'	H'_{max}	J' (H'/H'_{max})
Cleared and burned 2017	1.49	1.59	0.94
Cleared and Burned 2018	1.30	1.36	0.95
Cleared only	1.39	1.50	0.92
Intact chaparral	1.20	1.26	0.95

Vegetation assessment showed that intact chaparral had higher cover but lower species richness of both native and exotic species than all disturbed treatments (Figure 2) (cover: ANOVA: $F = 6.239$; df = 3, 6.36; $P = 0.026$) (number of species per m^2: Welch's t-test burned 2018 and intact: df = 2.386, $P = 0.029$; Welch's t-test cleared and intact: df = 9.665, $P = 0.029$). The density of non-native species was significantly higher in the burned 2017 treatment (Welch's t-test: df = 4.482, $P = 0.009$) and the cleared treatment (Welch's t-test: df = 9.346, $P < 0.001$) than in the intact treatment (ANOVA: $F = 15.927$; df = 3, 4.48; $P = 0.008$). The native cover in the intact treatment was higher than in all the disturbed treatments (burned 2017 [Welch's t-test: df = 5.779, $P = 0.056$], burned 2018 [Welch's t-test: df = 4.037, $P = 0.056$]), cleared treatment [Welch's t-test: df = 4.436, $P = 0.084$]).

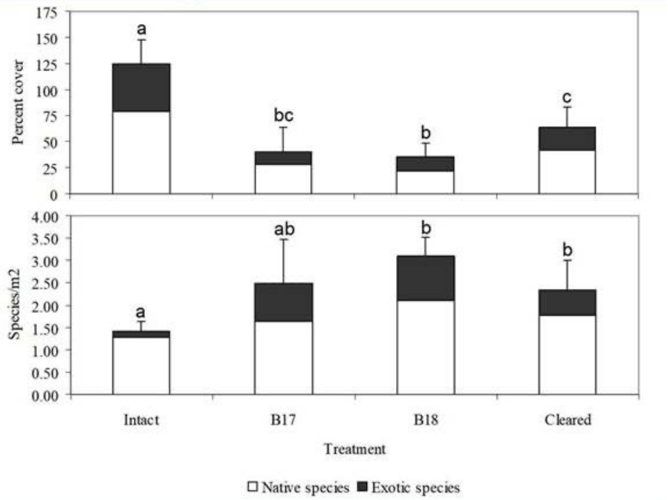

Figure 2. Upper chart: average percent cover; lower chart: average number of species per square m, with standard deviation of total cover and total species number; bars topped by a different letter are significantly different at P < 0.05. Bar segments show the contributions of native (clear) and exotic (black) species to the total for each of the four treatments (intact chaparral [Intact, 4 plots]), burned 2017 [B17, 5 plots], burned 2018 [B18, 3 plots], cleared only [Cleared, 8 plots], sampled at the Pine Hill unit of the Pine Hill Ecological Reserve, El Dorado County, California in May 2019.

Unvegetated bare ground, covered by soil, ash, or boulders accounted for an average of 61.76 ± 17.83% of the area for the burned treatments, while 36.10 ± 19.6% bare ground was found in the cleared treatments; intact chaparral had little bare ground (1.1 ± 2.2%) and over 100% total cover (average 124.48 ± 20.20%) by vegetation due to layers of low and high growing species (Table 5).

Examining cumulative percent cover by all exotic grasses (*Aira caryophyllea*, *Brachypodium distachyon*, *Bromus hordeaceus*, *Bromus madritensis rubens*, *Cynosurus echinatus*, *Festuca myuros*, *Gastridium phleoides*, *Hordeum murinum leporinum*), the cleared treatments had almost three times the cover by exotic grasses (22% cover) as the burned treatments (8% cover) and intact chaparral had 45% cover (Table 5). *Festuca myuros* and *B. distachyon* accounted for most of the exotic species cover in disturbed plots. There was but a single dominant exotic species in the chaparral, the grass *Brachypodium distachyon*. In only a single cleared treatment plot did this species reach a comparable cover of 60% but it was present in almost all disturbed treatment plots, cleared and burned.

Native and exotic annuals had generally low cover, bulbs had low but uniform cover throughout all treatments, and herbaceous perennials, like *Salvia sonomensis*, that were likely to be killed by fire had five times the cover in unburned treatments (cleared and intact) as in the fire treatments (2017 and 2018) (Table 5). Intact chaparral was dominated by a few native species with high cover; seeder shrubs *Arctostaphylos viscida* (17% cover) and CERO (8%), resprouting *Rhamnus ilicifolia* (8%), and *Frangula californica tomentella* (5%), and herbaceous *Salvia sonomensis* (14%) and *Carex xerophila* (14%). The cover of resprouting species increased 5-fold from earlier (2017) to later fire (2018) treatments primarily due to *Eriodictyon californicum*, but resprouters were low in cleared treatments. C*rocanthemum scoparium* was the dominant seeder in burned treatments, while mature CERO which by design escaped hand clearing, but is killed by fire, was the dominant seeder species in cleared treatments. *Arctostaphylos viscida*, cut down during clearing, did not resprout in the cleared treatment, and few seedlings were observed in the burned treatments.

There are two common species in the Lamiaceae family whose regeneration method following fire is little known. *Lepechinia calycina* has been reported to root-resprout following fire (Conrad 1987), but in our plant surveys elsewhere along the Pine Hill perimeter we saw resprouting only outside the fire's edge, with seedlings emerging from the ash within the burned areas. The regeneration strategy of *Salvia sonomensis* has not been reported. We found neither resprouts nor seedlings within burns; the low cover in fire treatments (2.5%) was entirely due to lateral stem expansion from outside the fire's edge. In contrast, the cleared treatment averaged 20% cover by this single species. Neither species is palatable to vertebrate herbivores (Conrad 1987) due to highly aromatic foliage.

The density of CERO seedlings was far lower in the cleared, unburned treatment (0.06 seedlings/m^2) than in either of the burned treatments (Welch's t-test, burned 2017 and cleared treatments: df = 9.182, $P < 0.001$; Welch's t-test, burned 2018 and cleared treatments: df = 5.018, $P = 0.038$) resulting in an overall significant treatment effect (ANOVA: $F = 81.41$; df = 2, 5.04; $P < 0.001$). There was no difference in seedling density between the 2017 (3.10/m^2) and 2018 (4.67/m^2) burned treatments assessed in either May (above) or October (October 2017: 4.44/m^2; 2018: 2.08/m^2). The increase in density in October 2019, in 2017 treatments was due to additional seedling germination after May 1, while the decrease in 2018 treatments was likely due to seedling mortality (2017: average of 24.11 ± 31.41% mortality versus 2018: average of 47 ± 44.83%, ns). Seedlings in the 2017 burned treatment grew taller during the 2019 growing season (May to October) than seedlings from the later 2018 burns (ANOVA: $F = 84.396$; df = 1, 203.55; $P < 0.001$).

Wildlife surveys

With the cameras, we collected an average of 343 ± 24 (SD) days of data per site. The cameras captured 2,872 unduplicated wildlife and human detections, with the most common species being mule deer, humans, rabbits, western grey squirrels, and grey foxes (Table 6).

Table 5. Cumulative percent cover of plants sorted into functional groups reflecting native or exotic status, life form, and method of recovery after fire (resprouting versus seeding) (averaged over treatments: burned 2017 [B17, 5 plots], burned 2018 [B18, 3 plots], cleared [8 plots], and intact [4 plots]).These data were collected from the Pine Hill unit of the Pine Hill Ecological Reserve, El Dorado County, California in May 2019.

Functional group	Cumulative percent cover (average [SD]) by treatment			
	2017	2018	Cleared	Intact
Exotic grass	7.8 (4.9)	8.8 (3.1)	21.9 (22.5)	45.1 (13.0)
Exotic annuals	5.1 (3.1)	5.1 (4.2)	2.3 (3.3)	0.0 (0)
Native grass	0.0 (0)	0.0 (0)	0.1 (0.4)	1.3 (2.5)
Native annuals	0.3 (0.4)	0.7 (0.6)	0.9 (1.1)	0.0 (0)
Bulbs	3.5 (4.0)	2.0 (1.0)	1.9 (1.5)	2.4 (1.6)
Herb perennials	3.2 (2.9)	7.1 (10.4)	23.0 (17.2)	30.3 (8.7)
Resprouters	14.5 (20.1)	3.4 (1.5)	4.0 (4.4)	16.3 (6.9)
Seeders	5.8 (2.9)	8.0 (10.4)	9.8 (7.9)	26.6 (24.0)
Unvegetated	59.9 (23.1)	64.9 (13.5)	36.1 (19.6)	1.1 (2.2)

Table 6. Species detected at 12 (nine treated and three control) camera sites in the Pine Hill unit of Pine Hill Ecological Reserve, El Dorado County, California from years 2018−2019. The average detection rates (DR, number individuals detected per camera per day) at control and treated camera sites is also shown.

Common name	Number detected	$\bar{x}$ ± SE DR control sites	$\bar{x}$ ± SE DR treated sites
Birds[a]	34	0.003 ± 0.003	0.008 ± 0.008
Opossum	4	0 ± 0	0.002 ± 0.004
Rabbits[b]	154	0.009 ± 0.012	0.052 ± 0.052
Grey squirrel	124	0.013 ± 0.016	0.037 ± 0.050
Human	360	0.008 ± 0	0.072 ± 0.094
Raccoon	8	0 ± 0	0.003 ± 0.004
Striped skunk	2	0 ± 0	0.001 ± 0.001
Black bear	12	0.003 ± 0.003	0.003 ± 0.004
Coyote	45	0.004 ± 0.002	0.012 ± 0.004
Domestic dog	13	0.001 ± 0.002	0.004 ± 0.004
Grey fox	96	0.008 ± 0.007	0.030 ± 0.044
Bobcat	46	0.005 ± 0.008	0.014 ± 0.020
Domestic cat	2	0 ± 0	0.001 ± 0.002
Puma	17	0.001 ± 0.002	0.005 ± 0.011
Mule deer	1,955	0.164 ± 0.143	0.544 ± 0.301
Total	2,872		

[a] All birds detected including California scrub jay, California towhee, western bluebird, wild turkey, and other species of the order Passeriformes.

[b] All rabbit species detected including cottontail rabbits and jackrabbits.

For western grey squirrel, rabbits, mule deer, grey fox, coyote, bobcat, and large carnivores (combined detections of black bears and pumas), we used single species occupancy models to analyze species' detection probabilities at each camera site.

Western grey squirrels' predicted detection probabilities at camera sites from the supported model (out of 25 tested) (Table 7) ranged from 0–0.297. Detection probabilities decreased in the spring ($p = 0$–0.086; $\beta = -0.708$, $P = 0.028$) and summer ($p = 0$–0.067; $\beta = -0.970$, $P < 0.01$) and increased in the winter ($p = 0$–0.297; $\beta = 0.808$, $P < 0.01$) compared to the fall. Further, their detection probabilities were positively correlated with detection rates of meso carnivores ($p = 0$–0.297; $\beta = 31.032$, $P < 0.001$) and negatively correlated with detection rates of large carnivores ($p = 0$–0.297; $\beta = -235.520$, $P < 0.001$). In the model with camera location as the only covariate, grey squirrel detection probabilities decreased at control camera sites compared to treated (burned and cleared) camera sites ($p = 0.022$; $\beta = -0.945$, $P = 0.001$) (Figure 3).

Table 7. Top-ranked single-species occupancy models and models with camera location tested to estimate the detection probabilities (*p*) of western grey squirrel, rabbits (cottontail and jackrabbits), mule deer, grey fox, coyote, bobcat, and large carnivores (pumas and black bears) at 12 camera sites in the Pine Hill unit of Pine Hill Ecological Reserve, El Dorado County, California from 2018–2019. We ranked models by the second order Akaike's Information Criterion (AIC_c) and considered models with a difference in AIC_c (ΔAIC_c) < 4 to be supported. We did not evaluate covariate effects on the occupancy (Ψ) of the sites by species. We also show the models with no covariates (.) as comparison, the number of model parameters (K), and the model weights (w_i).

Models by species[a]	K	AIC^{cb}	ΔAIC^{c}	w_i
Grey squirrel				
Ψ(.) *p*(season + camera location + meso carnivore DR + large carnivore DR)	8	835.04	0.00	1.00
Ψ(.) *p*(camera location)	3	909.70	74.66	0.00
Ψ(.) *p*(.)	2	919.19	84.14	0.00
Rabbits				
Ψ(.) *p*(property side + camera location + human DR)	7	948.34	0.00	0.92
Ψ(.) *p*(camera location)	3	1000.13	51.80	0.00
Ψ(.) *p*(.)	2	1023.45	75.11	0.00
Mule deer				
Ψ(.) *p*(camera location + property side)	6	3499.12	0.00	0.99
Ψ(.) *p*(camera location)	3	3546.90	47.78	0.00
Ψ(.) *p*(.)	2	3696.22	197.10	0.00
Grey fox				
Ψ(.) *p*(season + camera location + grey squirrel DR)	7	683.08	0.00	0.98
Ψ(.) *p*(camera location)	3	748.41	65.33	0.00
Ψ(.) *p*(.)	2	769.58	86.50	0.00
Coyote				
Ψ(.) *p*(ordinal day + camera location)	4	391.75	0.00	1.00
Ψ(.) *p*(camera location)	3	DNC		
Ψ(.) *p*(.)	2	DNC		

Models by species[a]	K	AICc[b]	ΔAICc	w_i
Bobcat				
$\Psi(.)$ p(property side + grey squirrel DR)	7	444.82	0.00	1.00
$\Psi(.)$ p(camera location)	3	480.14	35.33	0.00
$\Psi(.)$ $p(.)$	2	482.91	38.10	0.00
Large carnivore				
$\Psi(.)$ p(ordinal day + human DR)	4	225.49	0.00	0.39
$\Psi(.)$ p(ordinal day + meso carnivore DR)	4	228.31	2.82	0.09
$\Psi(.)$ p(ordinal day + camera location + human DR + mule deer DR)	5	228.93	3.44	0.07
$\Psi(.)$ p(ordinal day + human DR + mule deer DR)	5	228.93	3.44	0.07
$\Psi(.)$ p(ordinal day + camera location + human DR + meso carnivore DR)	5	229.20	3.71	0.06
$\Psi(.)$ p(ordinal day + human DR + meso carnivore DR)	5	229.20	3.71	0.06
$\Psi(.)$ p(ordinal day + camera location + human DR)	5	229.24	3.75	0.06
$\Psi(.)$ p(ordinal day + mule deer DR)	4	229.40	3.91	0.05
$\Psi(.)$ p(camera location)	3	DNC		
$\Psi(.)$ $p(.)$	2	DNC		

[a] DR = detection rate (defined as number of species detections at each camera divided by the number of total camera days), meso carnivores = combined detections of coyotes, bobcats, and grey foxes.

[b] DNC = the model did not converge.

Rabbit predicted detection probabilities at camera sites from the best supported model (out of 25; Table 7) ranged from 0–0.099. Compared to the eastern side, detection probabilities increased at camera sites located near the northern ($p=0.01$–0.083; $\beta=1.569$, $P<0.01$) and western ($p=0.035$–0.099; $\beta=1.786$, $P<0.001$) side of the property. Their detection rates were also negatively correlated with human detection rates ($p=0$–0.099; $\beta=-19.586$, $P<0.001$) and decreased at camera sites in unburned areas compared to treated areas ($p=0.004$–0.019; $\beta=-1.604$, $P<0.001$) (Figure 3).

Mule deer predicted detection probabilities from the top model (out of 19 tested) (Table 7) ranged from 0.066–0.361. Compared to the eastern side, detection probabilities decreased at camera sites near the northern ($p=0.093$–0.298; $\beta=-0.283$, $P=0.041$), southern ($p=0.066$–0.227; $\beta=-0.0634$, $P<0.001$), and western ($p=0.17$; $\beta=-1.014$, $P<0.001$) aspects of the property. Their detection rates also decreased at control camera sites compared to treated camera sites ($p=0.066$–0.093; $\beta=-1.424$, $P<0.001$) (Figure 3).

Grey fox predicted detection probabilities from the top model (out of 40) (Table 7) ranged from 0.003–0.188. Compared to fall, detection probabilities decreased in the spring ($p=0.003$–0.057; $\beta=-1.336$, $P<0.001$) and summer ($p=0.003$–0.066; $\beta=-1.185$, $P<0.001$). Their detection probabilities were positively correlated with western grey squirrel detection rates ($p=0.003$–0.188; $\beta=20.713$, $P<0.001$) (Figure 4). Grey fox detection rates also decreased in control camera sites compared to treated; this effect was not significant in the top model ($p=0.003$–0.02; $\beta=-0.24$, $P=0.561$), but it was significant when tested with no other covariates (Model with camera location only: $p=0.008$; $\beta=-1.326$, $P<0.001$) (Figure. 4).

Coyote predicted detection probabilities from the top model (Table 7) ranged from 0.002–0.018. Their detection probabilities decreased at camera sites located in control areas

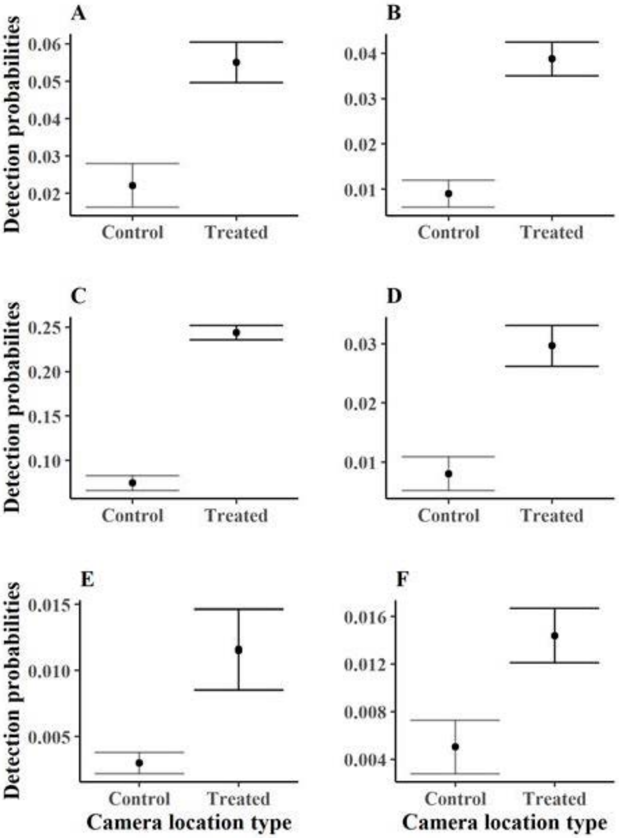

Figure 3. The effects of camera location in control (unburned and uncleared) areas and treated (burned and cleared) areas on predicted detection probabilities for grey squirrel (A), rabbits (cottontail and jackrabbits, B), mule deer (C), grey fox (D), coyote (E), and bobcat (F) from species' occupancy models with camera location as the only covariate for detection probabilities. The data was collected from 12 camera sites at the Pine Hill unit of the Pine Hill Ecological Reserve, El Dorado County, California from 2018−2019.

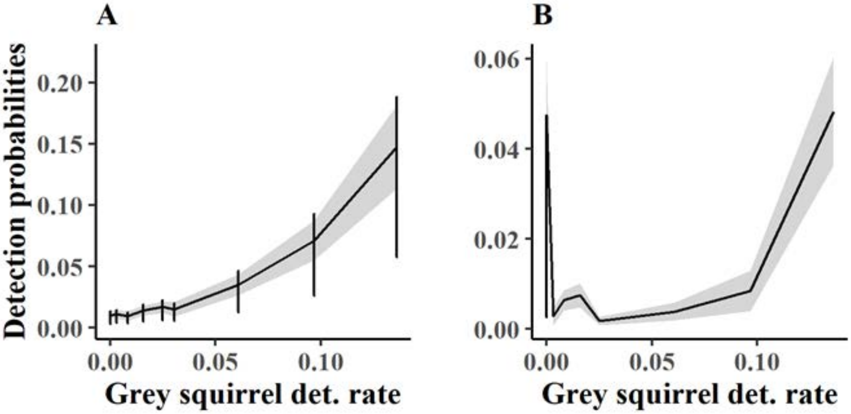

Figure 4. The effects of western grey squirrel detection (det.) rates on predicted detection probabilities from top occupancy models for grey fox (A; model with covariates season, camera location, and grey squirrel detection rate) and bobcat (B; model with covariates property side and grey squirrel detection rate). The data was collected from 12 camera sites at the Pine Hill unit of the Pine Hill Ecological Reserve, El Dorado County, California from 2018−2019.

compared to treated areas ($p=0.003$–0.02; $\beta=-1.36$, $P = 0.024$) (Figure 3). We compared a total of eight models with only temporal and location covariates (ordinal day, season, camera location, and property side) because all coyote occupancy models with species detection rates did not converge, likely due to the small sample size.

Bobcat predicted detection probabilities from the top model (out of 39 tested) ranged from 0.002–0.048. Compared to the eastern side, detection probabilities decreased at camera sites on the northern ($p=0.002$–0.008; $\beta=-3.853$, $P < 0.001$), southern ($p=0.005$–0.008; $\beta=-2.227$, $P < 0.001$), and western ($p=0.023$–0.048; $\beta=-2.92$, $P < 0.001$) sides of the property. Their detection probabilities were positively correlated with western grey squirrel detection rates ($p=0.002$–0.048; $\beta=21.633$, $P < 0.001$) (Fig. 4B). In the model with camera location as the only covariate, bobcat detection rates decreased at camera sites in control areas compared to treated areas ($p=0.005$; $\beta=-1.061$, $P = 0.026$) (Fig. 3F).

For large carnivores we compared a total of 47 models. Their predicted detection probabilities from the top models (Table 4) ranged from 0–0.018. Likely due to small sample sizes, large carnivore detection probabilities did not significantly correlate with any of the top covariates with a P-value < 0.05; however, there were some covariate effects in the top models that had a P-value < 0.1 reported here. Large carnivore detection probabilities were positively correlated with meso carnivore detection rates (model with ordinal day and meso carnivore detection rate: $p=0.003$–0.013; $\beta=5.6$, $P = 0.093$) and mule deer detection rates (model with ordinal day, human detection rate, and mule deer detection rate: $p=0$–0.018; $\beta=1.268$, $P = 0.089$). Large carnivore detection probabilities were negatively correlated with human detection rates (model with ordinal day, camera location, human detection rate, and mule deer detection rate: $p=0$–0.017; $\beta=-21.704$, $P = 0.094$). We also found that all tested models with camera location showed a decrease in large carnivore detection probabilities at camera sites located in control areas compared to treated areas; however, this effect was not significant (model with ordinal day, camera location, and human detection rate: $p=0.003$–0.005; $\beta=-0.714$, $P = 0.141$). The large carnivore occupancy model with camera location as the only covariate did not converge, likely due to the small sample size.

DISCUSSION

The establishment of a firebreak along the perimeter of Pine Hill presented an opportunity for collaboration between botanists from the California Native Plant Society and biologists from the California Department of Fish and Wildlife to study the effects of hand clearing and pile burning on vegetation growth and composition, and wildlife response to the disturbances. We found burning of many cleared vegetation piles may be a suitable substitute to allow fire cued seeds to germinate even if man-made fires may not follow natural burn regimes (Steel et al. 2015). Generally, we found that wildlife responded by using treated areas more than the control sites.

Plant survey

The distributions of the rare species on the Pine Hill perimeter were in accord with the vegetation community classification proposed by Wilson et al. (2009) and supported by extensive sampling by Gogol-Prokurat (2014). The three communities on the perimeter were oak woodland, where only GACAS and WYRE were found, and xeric to mesic chaparral, where all rare species except GACAS were found. Grassland was uncommon. The most mesic and shady northern aspect only contained the rare species FRDE and WYRE in a mixed chaparral-woodland community (Table 3).

Vegetation recovery and weed invasion. —Disturbance due to clearing only or clearing and burning resulted in the establishment of differing communities as measured by the Jaccard community coefficient, especially when determined using percent cover of shared species. This is an expected result, as such disturbances cause a niche opening for germination and growth of plants present in the seed bank, from vegetative structures, and of seeds dispersed into the opening. However, community difference was also found between treatment types, which suggested that the two treatments are not equal in terms of niche opening for different species. A concern is that a substantial proportion of plants differing between pairs of communities were exotic species, including species rated as moderately or highly invasive (California Invasive Plant Council 2020). Continued monitoring of the cleared perimeter of Pine Hill will be necessary to determine how colonization by invasive exotic species affects native community types.

Similarly, the high Shannon diversity found in the cleared and burned and cleared only treatments reflects the opening of niches for plants suppressed or barred from establishment in intact chaparral, which had the lowest Shannon diversity. Both establishment of native species and exotic species contributed to the high diversity of the treatments, with exotic species comprising 30% or more average cover in plots of the treatments. The historical fire disturbance in chaparral in California has allowed for successional communities that increase habitat variability; however, the high degree of invasion by exotic species may result in conversion of community type and lower native species diversity with changes in fire regime from historical patterns (Keeley and Davis 2007). The higher exotic species cover in the cleared treatment versus the cleared and burned treatment contributes to a different community type and to Shannon diversity, suggesting that continued clearing may cause conversion to a community type not historically found, with substantial cover by exotic species.

Disturbance (either clearing and burning or clearing only) resulted in colonization by a diversity of non-native plants. Further, there was higher cover by exotic grasses and native herbaceous perennials, species killed by fire, in the cleared than the burned treatments. However, intact, dense chaparral was no guarantee that weed invasion would not occur as the intact chaparral treatment contained high cover of *Brachypodium distachyon*, an exotic, fall germinating, shade tolerant grass. These last two traits mean that the grass germinates, grows up to 40 cm tall, and leaves dry, dead stalks under the shrub canopy after flowering in the spring (Piep 2012; CABI 2020). It is noted by CABI (2020) that this species can modify fire regimes, alter habitat, modify successional patterns, and form monocultures that smother native species resulting in reduced native biodiversity, including loss of rare species. Grasses provide the fuel to light the shrubs on fire (Brooks et al. 2004), and thus this one species may alter the natural fire dynamics of the chaparral by making invaded mature chaparral more vulnerable to fire. More generally, repeat fires, fueled by invasion of annual grasses, can lead to the conversion of deep-rooted shrub communities to shallow rooted grasslands and eliminate both resprouting and seeding native species (Keeley and Brennan 2012).

As well, *Brachypodium distachyon* was almost ubiquitous in the disturbed treatments, meaning clearing and fire were no impediment to its establishment. As the Pine Hill perimeter borders private property with a long history of disturbance and weed invasion (most homes were built before 1993, "Zillow" 2020), and the west perimeter contained a bulldozed fuel break wherein weed populations flourished (Merriam et al. 2006), seed sources are close by. In contrast, there was an almost complete lack of weeds 12 years after the Pondo 50-Palmer fire (pers. obs. Ayres and Meyer); this site borders areas that were little disturbed and/or lacked abundant weed populations to spread into this preserve. Whether our limited data, based on a single western aspect site, will hold true for other aspects and sites on Pine Hill is unknown, but we argue that invasion by *B. distachyon* is an unreported vulnerability for the gabbro chaparral, and by extension, other chaparral where it occurs.

In northern California chaparral, Potts and Stephens (2009) examined short term (three year) vegetation responses to fire and clearing (mastication) and found that mastication treatments had the highest number of exotic invasive species with 34% higher exotic annual grass abundance than the fire treatments, similar to what we found two years after clearing (39% higher grass cover in cleared relative to burned treatments). Comparisons of longer term outcomes (13 years post treatment) of fuel hazard reduction treatments suggest that mechanical removal (mastication) can result in higher numbers of non-native species, including grasses, than fire treatments, and that the chaparral becomes once more vulnerable to fire after only 10 years (Wilkin et al. 2017). Perchemlides et al. (2008) compared mastication to hand clearing plus pile burning in chaparral in southern Oregon 4–7 years after clearing and found no difference in the herbaceous community composition between the two types of clearing; further they posit that retention of shrub cover may retard invasion by exotic grasses even though the effectiveness of the fuel break may be diminished by leaving shrub cover. Merriam et al. (2006) found that proximity to fuel breaks, especially old ones constructed with bulldozers, was associated with non-native invasion as the fuel breaks were readily colonized by non-native species that then moved into nearby undisturbed vegetation. In addition to fuel break creation that minimizes bare ground, they recommend retaining some overstory canopy and ground cover to retard invasion by non-native plants. Our method of fuel break construction minimized bare ground, soil disturbance, and left low cover by CERO and *Salvia sonomensis* in place - all of which may retard erosion and deter the long-term establishment of non-native plants. In the cleared treatment, combined cover by these two species averaged 28% while in the burned treatments it was <4% indicating hand clearing was successful in retaining low cover. Whether this low cover results in a reduction of non-native species in the future is not known. Short and long-term monitoring of the Pine Hill perimeter will determine the fate of these fuel reduction treatments on vegetation recovery, weed invasion and CERO.

Ceanothus roderickii seedling germination and survival.—CERO is an obligate seeder (Boyd 2007); obligate seeders are killed by fire and persist as seeds in the soil until cued to germinate by the fire's heat or charred leachate; only the treatments that burned contained abundant seedlings. About 50% of the seedlings that germinated the first spring following fire died during the growing season. Seedling densities and survivorship range are in accord with that reported by Ayres(2011). In second-year treatments, about 25% of the crop of spring-marked seedlings, a mixture of new and 1-year old seedlings, died during the growing season but we observed new seedlings in the fall after our spring counts which indicated the window of germination extended beyond early spring and is greater than one year post-fire. Therefore, the assumption that most seeds cued to germinate by fire do so in the year following fire should be assessed for each species of concern. The taller stature of seedlings in the earlier 2017 burn treatment is consistent with these plants being one year older than those following the recent fires in 2018. Cleared treatments retained mature CERO plants that escaped hand clearing while there were no mature plants within the burned treatments.

We conclude that land managers in the short-term may consider that burn piles that promote fire cued seeds to germinate may be a viable substitute for widespread fires to establish fuel breaks where the risk to local homes and air quality concerns preclude large fires.

Wildlife survey

Our occupancy models for western grey squirrel, rabbits (cottontail and jackrabbits), mule deer, grey fox, coyote, and bobcat showed that these species all had a higher probability of being detected at cameras in treated areas (cleared and burned) compared to control areas (not cleared or burned). While this effect was not significant for large carnivores (black bears and pumas), likely due to a small sample size, large carnivores also showed higher detection probabilities overall at cameras placed in treated areas. We saw higher wildlife usage at the cleared perimeter of the property compared to the dense interior, suggesting that hand clearing and pile burning may contribute to higher wildlife activity. We hypothesize that this may be due to two main reasons. First, the unburned and uncleared areas were so dense that large bodied mammals such as pumas, black bears, and deer were not physically able to traverse them. Clearing the perimeter of Pine Hill allowed animals to travel the perimeter more easily. The habitat also likely became more attractive to species like mule deer and rabbits that prefer ecotones (Kufeld et al. 1988; Lombardi et al. 2003), since limited clearing created more habitat patchiness, which is supported by other studies in Arizona and California (Bock and Bock 1990). Second, openings in the canopy and reduced competition from shrubs allowed new vegetation to sprout, offering increased forage for herbivores along the fuel breaks. A seven-year study in Glenn and Colusa counties, California, demonstrated an increase in deer forage and consequently more usage and increased populations as a result of managed fire (Thornton 1982). Despite its high cover, intact chaparral mostly consisted of unpalatable native shrubs and *Salvia sonomensis* (Conrad 1987); in this study, we found that cleared and burned patches had 30-60% cover by soft, presumably palatable vegetation (Conrad 1987). Small-bodied herbivores, such as rabbits and squirrels which were not as limited by dense vegetation as larger mammals, had higher detection rates in treated areas and likely used increased food resources in these areas. Increased fuel break usage by herbivores may have contributed to higher detection rates of predators in treated areas, such as grey fox, which were also positively associated with higher detection rates of western grey squirrels. These findings are supported by camera detections of grey foxes carrying small prey species. Additionally, large carnivores appear to use the fuel breaks frequently, and mule deer detections had a positive association with large carnivore detection probabilities. Previous research in Southern California also reported that pumas used burned areas in response to prey availability (Jennings et al. 2016). Even though black bears were present in the study area, due to small sample sizes we could not identify whether they preferred treated areas; detection rates were the same in treated and control areas. Researchers in Florida did not report higher usage of burned habitats by black bears either (Stratman and Pelton 2007) and researchers in Arizona showed black bears avoided burned areas (Cunningham et al. 2003). Overall, generalizable responses of predators to burned areas appear to be lacking (Geary et al. 2019). Since we only evaluated one year of data in this study, it might be possible that effects were delayed due to generational lag time to produce foods for species such as bears (Young and Beecham 1986; Costello and Sage 1994). Further, comparing results to the general fire ecology literature base is complex because most studies focus on fire events at a much larger scale than in this study.

The property side where the cameras were placed was also an important covariate in the occupancy models. The east side of Pine Hill borders larger private properties and some include ponds which likely attract wildlife; both mule deer and bobcats had higher detection rates on the east side. On the south, west, and north sides more development is present, and mule deer and bobcats had lower detection rates in these areas.

Hand clearing and pile burning around the perimeter of Pine Hill appeared to have a positive effect on wildlife usage, and this could result in increased human-wildlife conflict in the housing developments surrounding the property. While we did not assess population sizes due to the limitations of the study area size and camera locations, we did observe an abundance of mule deer in the area and recorded higher detection rates in the fuel breaks. Previous research reported that burned areas may generate higher deer populations (Florence and Florence 1988), and even though burning has been utilized to improve deer habitat since at least the 1970s, responses might be more varied in other situations (Kie 1984; Klinger et al. 1989). While further study is needed to determine if the deer population is increasing at Pine Hill, higher deer activity at the perimeter of the property, adjacent to residential areas, as shown in our results could contribute to closer interactions between deer and humans than deer activity within the interior of Pine Hill. High mule deer activity near housing developments may result in higher rates of prized garden plant consumption, which can generate human-wildlife conflict in the area (McCullough et al. 1997; Rondeau and Conrad 2003). Anthropogenically created openings may persist as mule deer preferentially browse in these areas (Velamazán et al. 2018). Future research should focus on whether openings around the perimeter persist due to herbivory and may have positive effects on wildlife and special status plants.

Additionally, multiple pumas used the area, which will likely perpetuate and further generate conflict with hobby livestock such as goats and sheep in the area (Orlando 2008; Dellinger et al. 2020). From nearby property owner reports, we were previously aware of pumas predating on alpacas in the area and black bears damaging private property. Overall, even though Pine Hill is entirely landlocked by private properties, it appeared to support a wide assemblage of wildlife species including multiple carnivores and herbivores. This was contrary to our expectations as environments impacted by development are often considered as inadequate habitat to sustain large predators (Beier et al. 2008).

While Pine Hill is not open to public usage, our cameras detected significant usage of the fuel break by people due to increased connectivity around the property's perimeter; prior to hand clearing and pile burning it was not possible to walk the perimeter of the area because of dense shrubs. Increased public usage of Pine Hill could be detrimental to wildlife activity as human presence was negatively associated with usage by large carnivores (George and Crooks 2006; Wang et al. 2017). The area is not posted or signed indicating closure to the public, primarily because the area is difficult to access; however, we recommend signage be installed to deter public usage. People that used the area may all be neighboring property owners, but additional effort to understand who used the area is needed.

Epilogue

Pine Hill Flannelbush, *Fremontodendron decumbens*, is the rarest and most endangered plant in El Dorado County, only occurring on Pine Hill and in the immediate vicinity. Seedlings were last observed in December 1983 to April 1984 immediately following a 1 ha experimental fire on the top on Pine Hill in October of 1983 that burned down 38 mature flannelbush shrubs (Boyd 2001). Boyd (2001) marked and followed over 1100 seedlings for 13 months post-fire; 93% were killed by rodents, drought, or shading from flannelbush shrubs that resprouted from roots after the fire. Survival of seedlings was 2.4 times higher in the gaps between the former shrubs than under their canopies (11.5% vs 4.7) indicating the importance of dispersal away from mature shrubs.

On 4 May 2020, we found 4 seedlings within three fire scars from shrub piles that burned in March 2019 on the south perimeter. Shriveled seed leaves, cotyledons, were present on each seedling plant. The closest flannelbush shrubs were at least 6 m away. No seedlings were found in cleared and regrowing chaparral vegetation that was not burned as the seeds are cued to germinate by fire's heat (Boyd 2001). Ant dispersal is likely how the seeds got to the burn scars (Boyd 1996, 2001). The seeds have protein-rich elaiosomes attached to the tough-walled seed coat; native harvester ants, *Veromessor andrei*, can carry these seeds up to 12 m from the mature plant to their nests where they eat the elaiosomes and then deposit the seeds on their midden piles. The seedlings were large; spreading up to 13 cm, growing up to 8 cm tall, and with many true leaves suggesting they germinated with the 2019–2020 fall-winter rains and grew rapidly during the spring. There were likely many more seedlings that were eaten by rodents at the cotyledon stage (Boyd 2001). Will these new seedlings survive the summer drought and herbivores? We will see as we continue to monitor their survival and growth. Pile burning within 12 m of flannelbush shrubs may be an effective strategy to ensure the long-term survival of the species as its survival will depend on the establishment of new plants due to the eventual mortality of the mature shrubs (as yet unknown).

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Author Contributions

Conceived and designed the study: all authors
Collected the data: all authors
Performed the analysis of the data: MRC
Authored the manuscript: MK, MRC, DA
Provided critical revision of the manuscript: MK, MRC, DA, VM

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