Evaluating the Effects of Drought, Wildfire, and Habitat on Beetle Assemblages in a Mediterranean Oak Woodland

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ABSTRACT

In California, droughts are becoming more frequent and severe as a result of climate change. Additionally, wildfires in California are becoming larger and more intense due to forest management practices and changes in seasonal patterns. Wildfire and drought have the potential to negatively affect California ecosystems because they have the potential to drastically alter the physical and chemical characteristics of a habitat. Soil-dwelling insects such as beetles provide essential ecosystem services and can be used as bioindicators for ecosystem recovery. In 2013, a high intensity fire burned 3,700 acres on Mount Diablo in the San Francisco East Bay, California. To assess the response of invertebrate abundance and diversity, I used arthropod abundance data collected monthly at 6 sites; 3 in a burned area, and 3 in an unburned area. To measure habitat change, I calculated the Normalized Difference Vegetation Index (NDVI) of the area using aerial imagery. Fire and drought had a consistent significant impact on diversity and abundance of both families ($p < 0.05$). Vegetation change and habitat type had a significant impact on the abundance and diversity of Tenebrionidae ($p < 0.05$), but not Carabidae. These results show that the sampled species of beetles are adapted to fire but aren't well adapted to drought. Additionally, vegetation growth and habitat type show to be important to decomposers. These results have implications for forest management and climate action, as these decisions impact the severity of wildfires and droughts.

KEYWORDS

California, vegetation, NDVI, Carabidae, Tenebrionidae

INTRODUCTION

In California, climate change is increasing the occurrence and length of droughts and increasing the size and intensity of wildfires. Droughts are occurring more frequently and increasing in severity; the 2012–2016 California drought was the most severe California drought in the past 1200 years (IPCC 2013, Griffin and Anchukaitis 2014). Likewise, fire intensity and size are increasing because higher temperatures and seasonal shifts are leading to drier fuels and a longer fire season (Flannigan et al. 2008, Restaino 2013, Westerling 2016). Additionally, fire suppression, which has been practiced in the United States for over a century, has increased fuel loading in forests; both vertically and horizontally (Collins et al. 2019). In the case of the San Francisco East Bay Area, historic vegetation changes have created landscapes that have high fuel loads, and thus a greater risk for high intensity wildfires (Keeley 2005).

The impacts of severe drought and intense wildfires on California ecosystems can be monitored through ecosystem indicators such as vegetation and fauna. Change in habitat composition and structure can be measured directly, but ecological indicators provide information about the impacts of these changes on the biotics of an ecosystem (Pearce and Venier 2006). For a group of organisms to be good ecological indicators, they must be easy to sample and identify, functionally significant, and have a consistent response to disturbance (Pearce and Venier 2006). Beetles (Coleoptera) fulfill this role because they respond quickly to environmental changes, can be easily sampled and identified, and they are functionally significant (Longcore 2003). Specifically, ground beetles (Coleoptera: Carabidae) and darkling beetles (Coleoptera: Tenebrionidae) are easy to collect and identify in California, and they provide essential ecosystem services such as carbon and nitrogen nutrient cycling and are prey for other animals (Jansen 1997, Johnston 2000, Pressler 2019, Seastedt 1984,).

Beetle assemblages are impacted by precipitation, with higher abundance in the wet season than the dry season (Longcore 2003, Blanche et al. 2001). Number of individuals, species, and families is dependent on patterns of rainfall, both yearly and seasonally (Blanche et al. 2001). After a wildfire, post-fire beetle assemblages are dictated by the movement and survival of individuals, which is largely dependent on fire severity and habitat structure (Samu et al. 2010). Beetle recovery is affected by fire severity, because fire severity dictates mortality (Beaudry et al. 1997, Wikars 1995, Wikars 1997). High intensity wildfires disturb the habitat of soil-dwelling beetles by killing understory vegetation, burning the leaf litter layer, and burning the top layer of soil (Gongalsky et al. 2012). Wildfires remove organic material and nutrients from the soil, so post-fire substrate is not favorable for the decomposer prey organisms because much of the material is already decomposed from the fire (Certini 2005, Gongalsky et al. 2012). Post-fire nutrient-poor soil is unfavorable to soil-dwelling decomposers until vegetation regrows and contributes leaves and other debris to the substrate (Malmström 2010).

Ecosystems in California have evolved in a Mediterranean climate with periods of drought and instances of wildfires; however, they evolved with less severe droughts and less intense and more frequent wildfires (Collins et al. 2019). Beetles in California ecosystems may be adapted to fire, and their populations decline in the absence of fire (Andersen and Muller 2000), negatively affected by intense burning (Gongalsky et al. 2012), or unaffected by burning, regardless of fire intensity (Pressler 2019). In 2013, a high intensity fire burned 3,700 acres on Mount Diablo in the East Bay Area. This area is an oak woodland ecosystem dominated by Grey Pine and Blue Oak. Historically, before Euro-American settlement, fires occurred in forests about every 7-12 years (Van de Water and Safford 2011). This area had not been burned for fuel management in 15 years, which caused a build up of fuels and contributed to the intensity of the fire (Cuff and Nardi 2013). California was in a drought when the Morgan fire happened, and the drought continued into the post-fire period. It isn't clear how the interactions between the fire and the drought impacted the local ecosystem.

In this project, I am to understand how soil-dwelling beetle assemblages are impacted by severe drought and intense wildfire, in order to understand the impacts of climate change on California oak woodland ecosystems. My objectives are to 1) determine if drought impacts the abundance and diversity of beetles, 2) evaluate the effects of fire on beetle abundance and diversity, and 3) determine if there is a relationship between vegetation regrowth and beetle abundance and diversity.

METHODS

Study system

This study is focused on Perkins Canyon in Mount Diablo State Park, just outside the town of Perkins Canyon, CA (37.894157°N, -121.875009°W) in Contra Costa County. This area is an oak woodland dominated by Blue Oak (*Quercus douglasii*), Grey Pine (*Pinus sabiniana*), and Interior Live Oak (*Quercus wislizeni*). This canyon was burned during the Morgan Fire in September 8th, 2013, which burned 2,300 acres in Mount Diablo State Park (Figure 1).

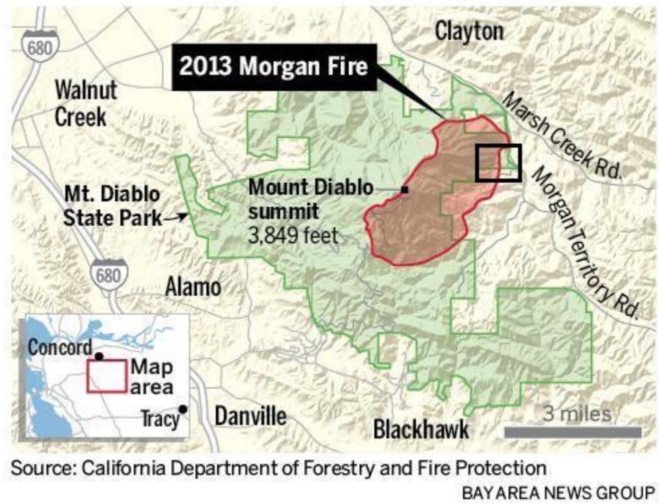

Figure 1. Morgan fire area in Mount Diablo State Park. Morgan fire area is in red, area of sample collection is in black box.

Environmental data

The precipitation data I used was collected at the Marsh Creek Fire Station in Contra Costa County (37.8949°N,-121.8635°W). The data was downloaded from the California Data Exchange Center (California Department of Resources 2020) on February 10th, 2020 for the date range of January 1st, 2014 to December 31st, 2019. Downloaded data includes hourly measurements of precipitation measured using a tipping bucket rain gauge.

Vegetation surveys

To assess changes in tree presence and growth, Heath Bartosh, Kipling Will, and Alyssa Zhang conducted an initial tree survey in 2015 and I, along with Kipling Will, Ursula Harwood, and Gavin Williams conducted a second survey on November 22nd, 2019. These surveys included measuring diameters at breast height (DBH) for all trees within 5 meters of each pitfall trap, recording the species of the trees, and recording whether the tree was living or dead. I calculated the basal area (BA), which is the sum of the cross-sectional area at breast height of all trees, for the burned and unburned sites, and summarized species composition and tree mortality.

Remote sensing

To determine post-fire vegetation recovery, I developed a maximum buffer distance by entering the coordinates of all 6 sites into ArcMap 10.7. I buffered these points 35.815 meter so that all the sites had the same buffer distance but the buffers around each site were never overlapping with another. I clipped these buffered sites to the boundary of the Morgan Fire. To assess NDVI at all of the sites, I used aerial imagery obtained from Planet Labs (Planet Team 2017). I used 4-band PlanetScope Scene imagery, with a resolution of 3-4 meters, and a cloud cover of less than 5%. I used the surface reflectance imagery, which has been corrected for aerial reflectance. I used one aerial image from each month within 7 days of when the invertebrate traps were put out. Choice of date depended on availability of imagery that had cloud cover under 5% and 3-4 meter resolution. I used ArcMap 10.7 to evaluate the Normalized Difference Vegetation Index (NDVI) of all six sites each month (ArcMap 2019). NDVI is calculated using the near-infrared band and the red band of each aerial image.

$$\text{NDVI} = \frac{(\text{NIR} - \text{Red})}{(\text{NIR} + \text{Red})}$$

NDVI can be used to assess vegetation recovery because it is a way to identify the presence of living vegetation in an area from an aerial image. I analyzed each image, and obtained an NDVI value for each site each month from July 2014-July 2019; I analyzed a total of 52 images.

Arthropod Sampling

To assess invertebrate abundance and diversity, I used arthropod data collected by Professor Kipling Will and others from the Will Lab at the University of California, Berkeley. Will established 6 sites in the canyon, 3 in the burned area and 3 in the unburned area (Figure 2a and Figure 2b). The sites are in two different habitat types: oak and pine woodland, and open oak grassland (Figure 3). Will used a total of 10 traps placed in a line transect at each site, with 5 meters between each trap. Pitfall traps were opened once a month around the time of the new moon, they were open for 3-4 nights, and then the 10 cups were collected and combined; after each collection there were a total of 6 samples. These samples were sorted in the laboratory, and organisms were identified to at least genus, and for some taxa to species. Using pitfall traps to measure abundance presents the possibility of confounding abundance and activity; however, that bias is consistent across all samples (Hoekman et al. 2017). Additionally, samples gathered with pitfall traps reliably reflect the population metrics of arthropod species in California oak woodlands (Weary et al. 2019).

(a) **(b)**

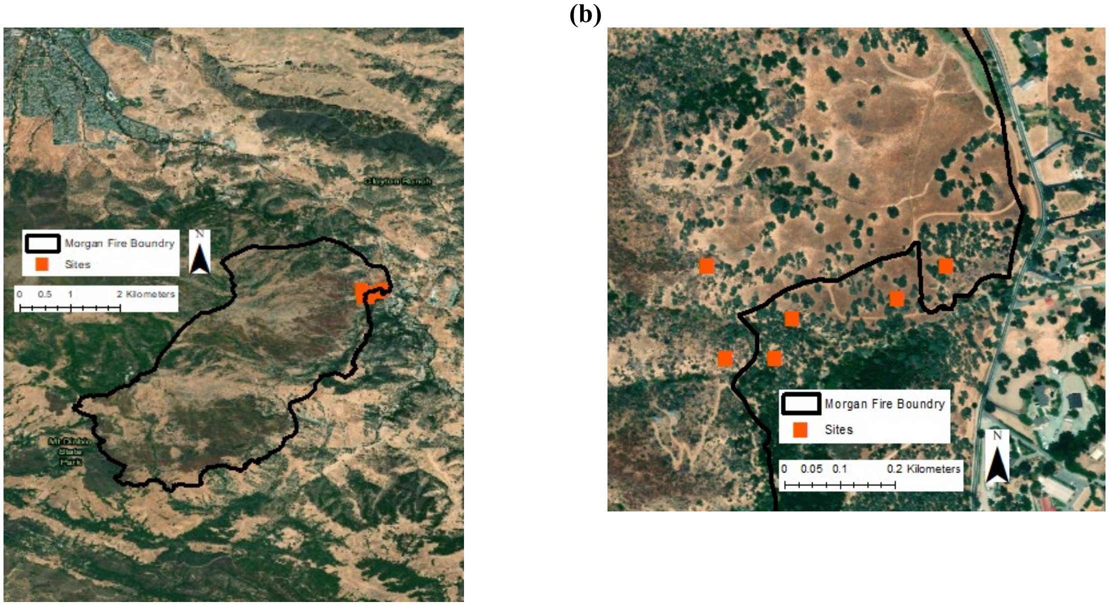

Figure 2. Aerial imagery with Morgan fire boundary and collection sites. Three sites are located inside the fire boundary, and three are located outside the boundary.

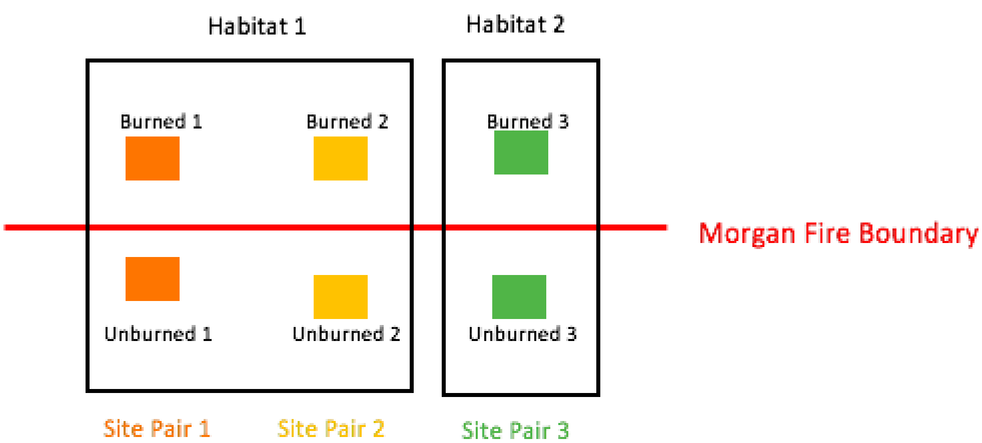

Figure 3. Study Design. There are 6 sites, 2 treatments, and 2 habitat types. Habitat 1 is oak and pine woodland, and habitat 2 is open oak grassland.

Analysis

Diversity and abundance analysis

To calculate abundance of beetles, I summed the abundances of all species in the group or family. To calculate the diversity of beetles in each sample, I used the Shannon Diversity Index as detailed in Daly et al. (2018). This index has been used in other studies to generally measure insect diversity (Ghani and Maalik 2020), and to measure beetle diversity specifically (Lencinas et al. 2019).

$$H_{\mathrm{Sh}}(\mathbf{p}) = -\sum_{i=1}^{S} p_i \ln(p_i)$$

$$p_i = \frac{\text{total number of individuals of species } i}{\text{total number of individuals of all species}}$$

Regression analysis

To understand the vegetation interactions occurring in the ecosystem, I performed a standard multivariable regression analysis in R studio (RStudio Team 2019). Additionally, to determine the effects of drought, wildfire, vegetation growth, habitat, and season on abundance and diversity of beetles, I performed 6 standard multivariable regression analyses in R studio (RStudio Team 2019).

Definitions of regression variables. I defined abundance as the number of individuals collected (total abundance or family abundance). Wildfire is whether the sample was collected in a burned area or unburned area (burned = 1, unburned = 0). Diversity is the Shannon Diversity index score. Drought is whether the sample was collected before or after April 7th, 2017 (drought = 1, no drought = 0). Season is whether the sample was collected in the wet season or the dry season (wet season = 1, dry season = 0). The wet season is defined as November - April, and the dry season is May - October. Habitat is whether the sample was collected in the oak and pine woodland habitat or the open oak grassland habitat (oak and pine woodland = 1, open oak grassland = 0). NDVI is the NDVI of each site for each sample date.

Regression Equations. I carried out seven different regressions to compare responses to the physical environment. I didn't include the precipitation data because the timing of beetle response to precipitation isn't fully understood. Also, it is not clear whether beetles respond to the length or intensity of a precipitation event. Because of these factors, adding in precipitation as a predictor may have caused inaccuracies.

NDVI ~ Drought + Treatment + Season + Habitat

Total Abundance ~ Drought + Wildfire + Season + Habitat + NDVI

Carabid Abundance ~ Drought + Wildfire + Season + Habitat + NDVI

Tenebrionid Abundance ~ Drought + Wildfire + Season + Habitat + NDVI

Total Diversity ~ Drought + Wildfire + Season + Habitat + NDVI

Carabid Diversity ~ Drought + Wildfire + Season + Habitat + NDVI

Tenebrionid Diversity ~ Drought + Wildfire + Season + Habitat + NDVI

RESULTS

Physical habitat: precipitation

Over the five-year period, the average yearly rainfall was 24.62 (± 7.81) inches, and the average monthly rainfall was 2.05 (± 3.02) inches. The year with the lowest rainfall was 2015 with 12.39 (± 1.40) inches, and 2017 had the highest rainfall with 34.34 (± 4.59) inches. All years follow generally the same pattern, with monthly precipitation peaking December to March, and lowest monthly precipitation occurring July to September (Figure 4). The California drought occurred from 2011 to 2017, and California was taken out of the drought emergency on April 7th, 2017.

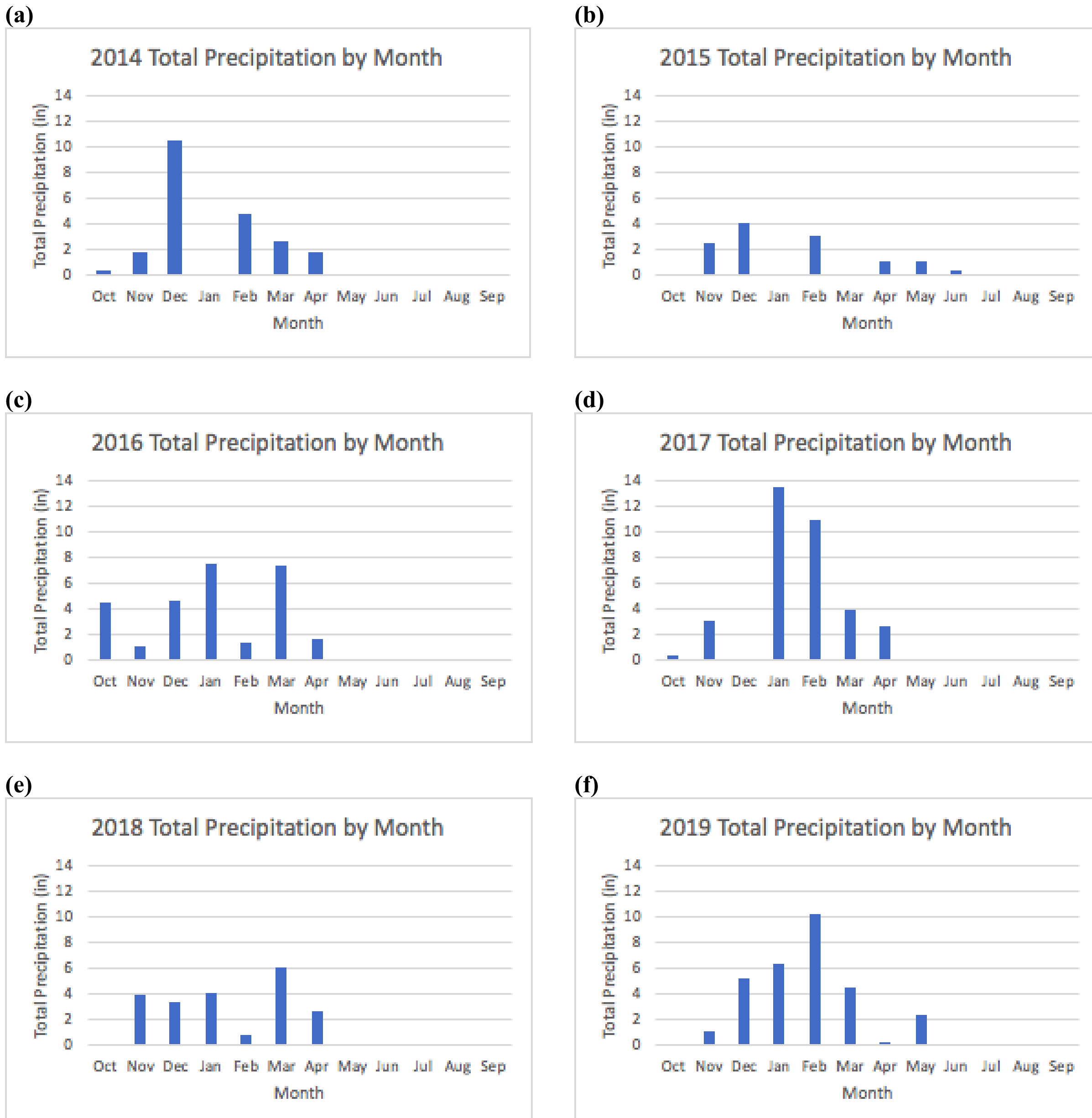

Figure 4. Total precipitation by month and water year for 2014-2019. Data were collected at Marsh Creek Weather Station (37.8949°N,-121.8635°W), and acquired from the California Data Exchange Center website. Precipitation was measured using a rain tip gauge. Each graph is a water year (October - September).

Physical habitat: vegetation

The species present in both vegetation surveys were Blue Oak (*Quercus douglasii*), Grey Pine (*Pinus sabiniana*), and Interior Live Oak (*Quercus wislizeni*) (Table 1). In 2015, 83 trees were surveyed in the burned sites. Of these 83 trees, 82% were dead, and 18% were living. In

2019, 90% of the trees in the burned sites were dead, and 10% of trees were living. Of the dead trees in 2019, 49% of trees were dead and down, 41% of trees were dead and standing. The trees around the traps in the habitat 1 burned sites had 100% tree mortality due to the wildfire, while the trees around the traps in the habitat 2 burned site had 43% tree mortality (Figure 5). Basal area was decreased by 947 ft^2 per acre, after the wildfire for site pair 1, and 809 ft^2 per acre for site pair 2. Site pair three basal area decreased 281 ft^2 per acre after the burn (Figure 6). In 2019, the two unburned sites in habitat 1 had between 20-40 trees within the 5 meter radius of the traps, while habitat 2 had 14 trees per site. Habitat 1 is characterized by more Grey Pine and Interior Live Oak, higher tree density, and higher tree mortality leading to a larger decrease in basal area. Habitat 2 has primarily Blue Oak, lower tree density, and lower tree mortality leading to a smaller decrease in basal area. Tree regeneration, in the form of Grey Pine seedlings and saplings, was found at 2 the burned sites and 2 unburned sites in habitat 1 (Table 2). No regeneration was found in habitat 2.

Table 1. Tree Species by percentage in 2015.

Site	Blue Oak	Grey Pine	Interior Live Oak
Habitat 1	40%	43%	17%
Habitat 2	82%	13%	3%

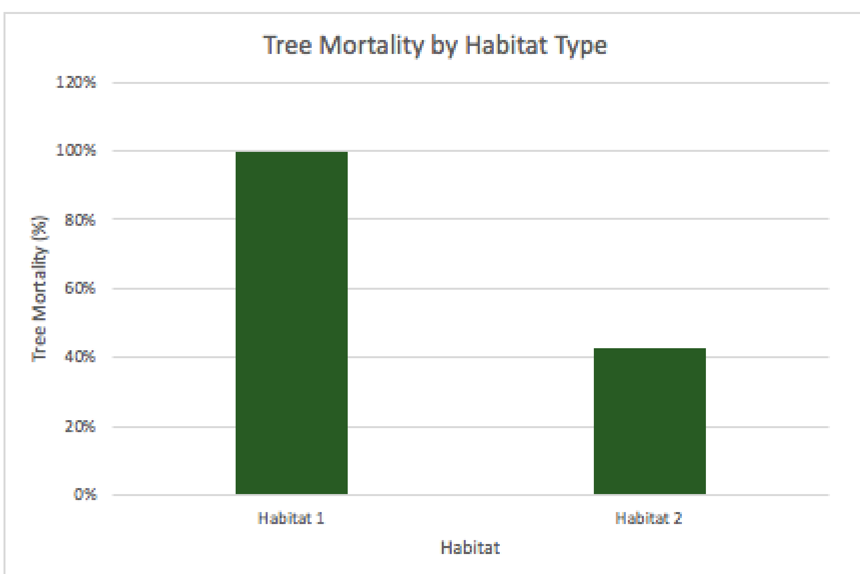

Figure 5. Tree mortality by habitat type. All trees were recorded that fall within a 5 meter radius of a trap.

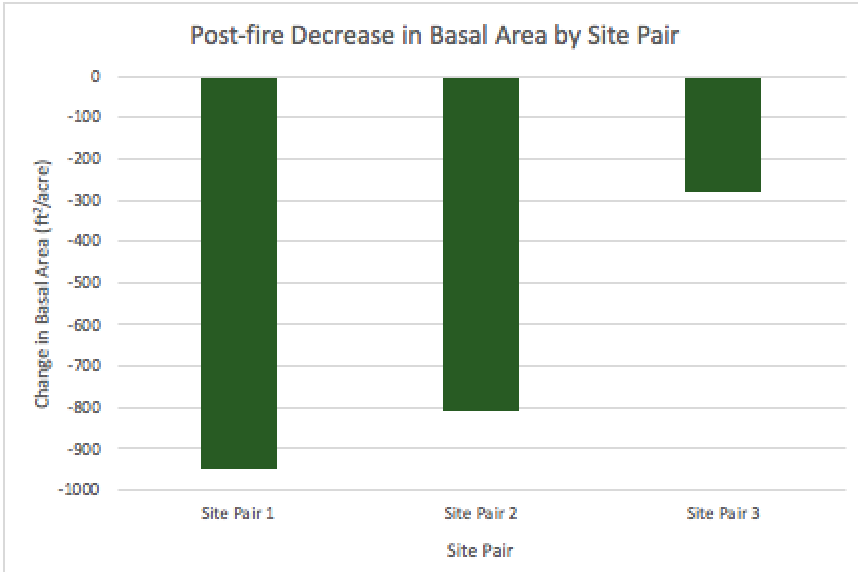

Figure 6. Post-fire decrease in basal area by site pair. Site pair 1 and 2 are in Habitat 1, and site pair 3 is in Habitat 2.

Table 2. Tree regeneration measured in 2019.

Site	**Species**	**Height**			**DBH (if height greater than 4.5 feet)**	**Total**
		<1 foot	**1-3 feet**	**3-4.5 feet**	**0-1 in DBH**	
Burned 1	Grey Pine	1	3	1	4	9
Unburned 1	Grey Pine	0	1	1	0	2
Burned 2	Grey Pine	0	0	5	3	8
Unburned 2	Grey Pine	2	1	2	2	7

NDVI

NDVI of all the sites, and thus vegetation growth and decay, generally follow the same seasonal pattern. NDVI for all sites peaks in March to April and is lowest from September to November. The average NDVI of the burned sites was 0.30, with a range from -0.02 to 0.64. The average NDVI of the unburned sites was 0.37, with a range from 0.07 to 0.65. The difference in

NDVI between the burned and unburned sites in site pairs 1 and 2 is much larger and more varied, while the difference in NDVI between the two sites in site pair 3 is not as large and less varied (Figure 7). The average difference in NDVI between the burned and unburned sites for pair one and two was -0.094 and -0.12 respectively. For pair three, the average difference was 0.0045.

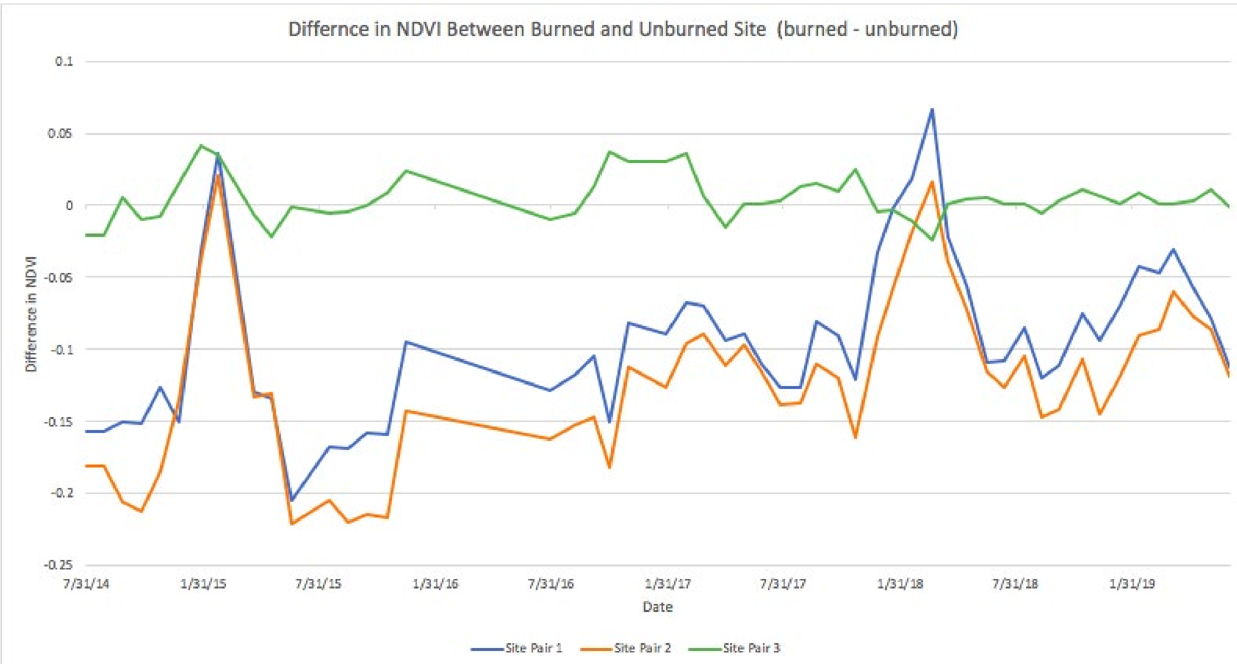

Figure 7. Difference in NDVI between burned and unburned sites in each site pair. Negative values indicate that the unburned sites had higher NDVI values.

Abundance and diversity of sampled invertebrates

A total of 1337 individuals from the Carabidae and Tenebrionidae families were sampled from the 6 sites in the 5 year period. A total of 694 Tenebrionids and 643 Carabids were collected. Those individuals represented 28 species, and 20 genera, and 2 families. The most abundant species was *Pterostichus californicus*, the most abundant genera were *Pterostichus* and *Eleodes* (Table 3).

Table 3. Total sampling abundance by species and treatment type. Sampling occurred once every month at three burned sites and three unburned sites.

Genus	Species	Total Abundance	Total Abundance	Total Abundance
Pterostichus	*californicus*	297	211	86
Pterostichus	*protensiformis*	67	14	53
Pterostichus	*angustus*	30	18	12
Pterostichus	*vicinus*	107	53	54
Amara	*rectangula*	17	10	7
Amara	*conflata*	32	27	5
Scaphinotus	*interruptus*	4	0	4
Scaphinotus	*striatopunctatus*	20	6	14
Promecognathus	*laevissimus*	27	19	8
Calathus	*ruficollis*	22	20	2
Dichierus	*piceus*	15	10	5
Axinopalpus	*biplagiatus*	5	4	1
Notiophilus	*semiopacus*	4	3	1
Coniontus	*sp.*	176	102	74
Eleodes	*dentipes*	122	91	31
Eleodes	*constrictus*	106	57	49
Nyctoporis	*aequicollis*	151	96	55
Coelocnemis	*magma*	43	32	11
Corticeus	*praetermis*	1	1	0
Blapstinus	*discolor*	1	0	1
Blapstinus	*dispar*	12	9	3
Alaudes	*singularis*	1	1	0
Apocrypha	*anthicoides*	38	10	28
Apsena	*rufipes rufipes*	21	7	14
Apsena	*pubescens*	5	0	5
Cibdelis	*blaschkei*	5	1	4

Phloeodes	*diabolicus*	6	2	4
Phloedes	*plicatus*	6	3	3

The difference in abundance of Carabids between burned and unburned sites in Habitat 2 overall shows less oscillation, and less total abundance (Figure 8). In Habitat 1, oscillations were larger and total abundance higher. Tenebrionids show a similar pattern (Figure 9). For total diversity, there is a similar trend; low intensity fire promotes a consistently higher diversity, while high intensity fires result in larger oscillations in diversity and higher total diversity overall (Figure 10).

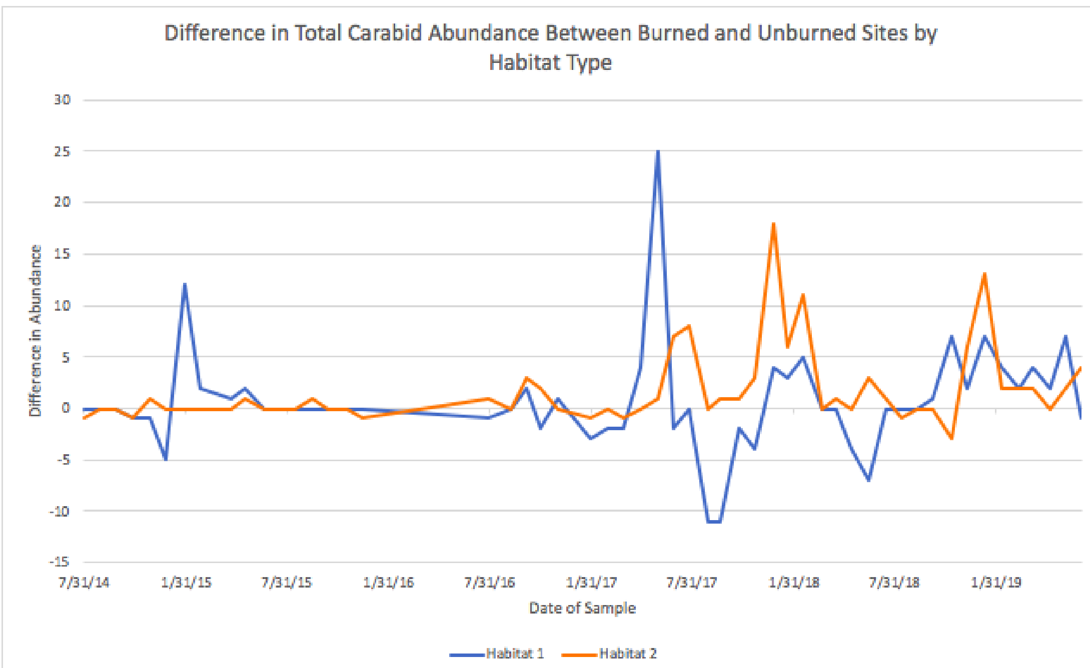

Figure 8. Difference in carabid abundance between burned and unburned site by habitat type. This was calculated by subtracting the unburned total abundance from the burned total abundance.

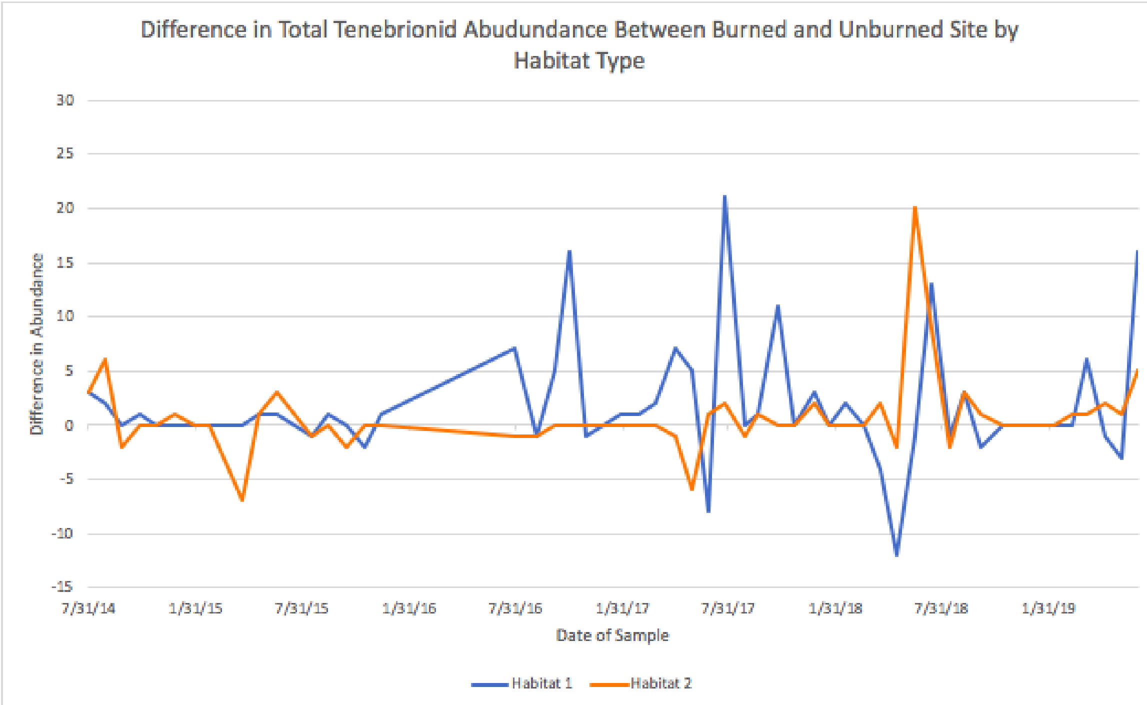

Figure 9. Difference in tenebrionid abundance between burned and unburned sites by habitat type. This was calculated by subtracting the unburned total abundance from the burned total abundance.

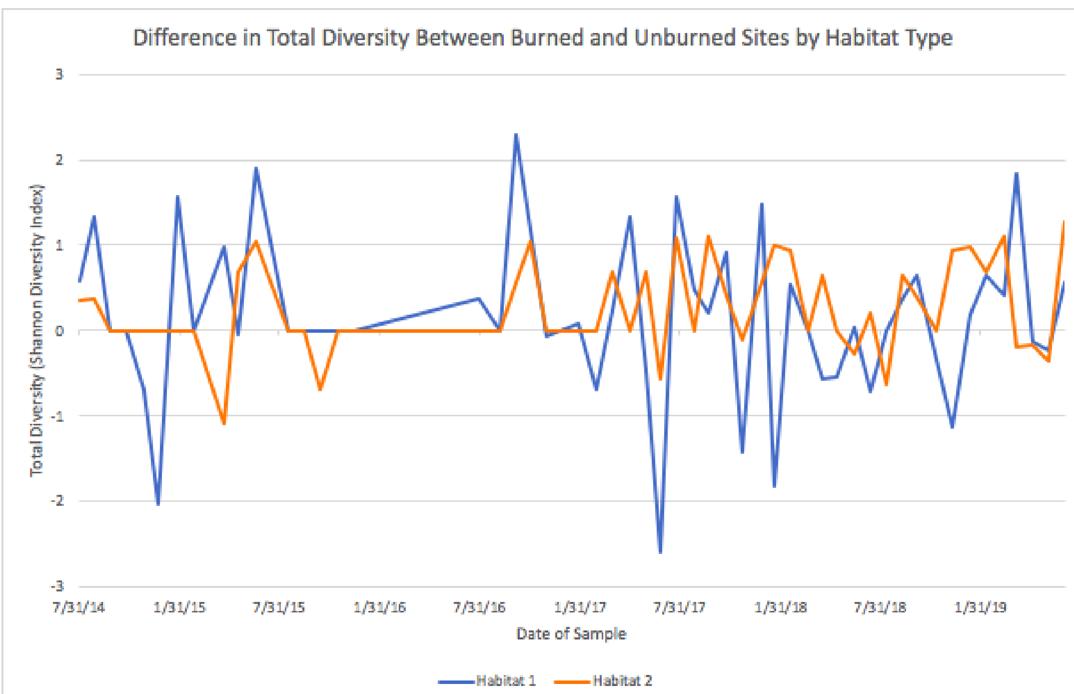

Figure 10. Difference in total diversity between burned and unburned sites by habitat type. This was calculated by subtracting the unburned total diversity from the burned total diversity**.**

Regression analysis

Vegetation Regression Model

The regression model for vegetation significantly predicts NDVI ($p < 0.001$) (Table 4). NDVI is significantly predicted by drought, treatment, and season. Drought and wildfire have a negative impact on NDVI, while wet season has a positive impact (Table 4).

Table 4. Regression coefficients for vegetation model.

	Regression Coefficients				
	Drought	**Wildfire**	**Season**	**Habitat**	**p-value of model**
NDVI	-0.189678*** (0.013483)	-0.071252*** (0.013443)	0.106600*** (0.013483)	-0.003563 (0.014258)	<0.001

P value: ^ = 0.05, * = 0.01, ** = 0.001, *** = <0.001

Abundance Regression Models

The regression models for total abundance, carabid abundance, and tenebrionid abundance significantly predict the dependent variable ($p < 0.001$) (Table 5). Drought, wildfire, and season are significant predictors for total abundance, while habitat and NDVI are not. Drought had a negative impact on abundance, and wildfire had a positive impact. The presence of drought causes a decrease in total abundance of 3.30 individuals, on average. Carabid abundance and Tenebrionid abundance are decreased by 2.10 and 1.19 on average, respectively. Wildfire increases total abundance, carabid abundance, and tenebrionid abundance by 2.02, 0.83, and 1.14 on average, respectively. Season affected abundance, but the effect varies between the families. Tenebrionid abundance is lower in the wet season, while Carabids abundance is higher in the wet season. Habitat and NDVI were not significant predictors for abundance.

Diversity Regression Models

The regression models for total diversity, carabid diversity, and tenebrionid diversity significantly predict the dependent variable (p < 0.001) (Table 5). Total diversity and tenebrionid diversity are significantly predicted by drought, wildfire, season, habitat and NDVI. Carabid abundance is significantly predicted by drought, wildfire, and season, while carabid diversity is significantly predicted by only drought and season. Drought has a negative effect on diversity, while wildfire has a positive effect. Drought decreases total diversity, carabid diversity, and tenebrionid diversity by 0.31, 0.14, and 0.17 on average, respectively. Wildfire increases total diversity and tenebrionid diversity by 0.19 and 0.13 on average, respectively. Tenebrionid diversity is lower in the wet season, while carabid diversity is higher in the wet season. Habitat and NDVI increase total diversity and tenebrionid diversity. Habitat increases total diversity by 0.14 on average, and tenebrionid diversity by 0.13 on average. NDVI increases total diversity by 0.75 on average, and tenebrionid diversity by 0.38 on average.

Table 5. Regression coefficients for beetle population metrics.

Beetle population metric	**Regression Coefficients**					
	Drought	**Wildfire**	**Season**	**Habitat**	**NDVI**	**p-value of model**
Total Abundance	-3.3021*** (0.6488)	2.0249*** (0.5269)	-2.2520*** (0.5550)	0.7875 (0.5350)	3.7687^ (2.1413)	<0.001
Carabid Abundance	-2.101897*** (0.445069)	0.834045* (0.361502)	1.061809** (0.380746)	0.230805 (0.367043)	0.009992 (1.469037)	<0.001
Tenebrionid Abundance	-1.1917* (0.4725)	1.1356** (0.3847)	-3.2555*** (0.4052)	0.5236 (0.3900)	3.5908* (1.5617)	<0.001
Total Diversity	-0.30742*** (0.07312)	0.19180** (0.05939)	-0.25077*** (0.06256)	0.13843* (0.06030)	0.74601** (0.24136)	<0.001

Carabid Diversity	-0.14433** (0.05139)	0.04054 (0.04174)	0.19479*** (0.04397)	0.03047 (0.04238)	0.19033 (0.16964)	<0.001
Tenebrionid Diversity	-0.17422** (0.05621)	0.12742** (0.04565)	-0.42123*** (0.04808)	0.13435** (0.04635)	0.38085* (0.18553)	<0.001

P value: ^ = 0.05, * = 0.01, ** = 0.001, *** = <0.001

DISCUSSION

Drought, wildfire, and vegetation are all relevant components of California ecosystems, and shape the overall habitat. Beetles are linked to their environment, and they are sensitive to habitat change. Both the carabids and tenebrionids that were sampled were negatively impacted by drought. Wildfire has a positive effect on carabids and tenebrionids due to the ability of fire to create new microhabitats and make certain nutrients more available (Whelan 1995). Beetle populations vary by season, and vegetation regrowth and habitat significantly impact sampled tenebrionids, but not carabids. Tenebrionids are primarily decomposers, so they rely on vegetation to provide leaf litter, and may have a more direct need for vegetation in their habitat. Carabids are primarily predators and scavengers, so their populations are typically linked to the presence of prey arthropods. Overall, the tenebrionids and carabids that were sampled respond significantly to changes caused by drought, wildfire, and season; however, responses to habitat difference and vegetation recovery are affected by life history traits.

Drought

Drought negatively impacts abundance and diversity of carabids and tenebrionids independent of habitat type or treatment. Reduction in beetle abundance and diversity in response to drought may result from reduced soil moisture, reduced litter mass, and a change in soil chemistry due to reduced precipitation (Williams et al. 2014). Beetle abundance is significantly reduced in dry plots compared to wet plots, and beetle abundance and richness are likewise positively related to soil moisture (Williams et al. 2014). Additionally, carabids are negatively impacted by changes in litter moisture (Yi and Moldenke 2005) whereby precipitation levels affect beetle abundance by altering microhabitat. Essential microhabitat for soil-dwelling forest invertebrates is created by biotic processes that are affected by water availability (Chikoski et al. 2006). Examples of these biotic processes are decomposition rates, and the abundance of microinvertebrates that influence decomposition and are prey species for beetles (Johnston 2000).

Wildfire

Wildfire has a positive impact on abundance and diversity for both the tenebrionids and carabids that were sampled. Beetles in California evolved in a historical fire regime where wildfires were less intense and more frequent (Collins et al. 2019). Beetles populations decline in the absence of fire, likely because they are adapted to wildfire (Andersen and Muller 2000). In contrast to studies where intense fires have a negative impact on beetle abundance (Gongalsky et al. 2012), fire in the oak and pine woodland habitat had high tree mortality but did not have declines in beetle abundance. This difference in response may result from the period of sampling, as high intensity fires can initially cause high mortality of soil-dwelling invertebrates, but populations of soil invertebrates are expected to recover over time (Holliday 1992).

Graphical differences in the pattern of recovery of the two habitat types can be used as a way to understand how beetles respond to different fire intensity. The difference in abundance between burned and unburned sites in Habitat 2 shows less overall oscillation, and lower total abundance. In Habitat 1, there are larger oscillations and higher total abundance. For the Morgan fire, lower intensity fires consistently promote higher abundance, while high intensity fires result in more oscillations in abundance, but higher total abundance. For total diversity, a similar trend occurs; low intensity fire promotes a consistently higher diversity, while high intensity fires result in larger oscillations in diversity and higher total diversity.

These findings suggest that beetle populations in a high intensity burned area may be more sensitive to seasonal changes. This may be linked to vegetation, as the NDVI of Habitat 1 shows larger seasonal change than the NDVI of Habitat 2. This seasonal vegetation growth likely impacts soil habitat by providing leaf litter and shade, which is otherwise absent due to high tree mortality in Habitat 1. Beetles may move into the area that has the most suitable habitat in any given season. Fire severity is an important factor explaining differences in responses to fire for soil-dwelling organisms, and recovery patterns can differ based on severity of the burning (Malmström 2010).

Although this graphical difference is apparent, Habitat type was not a significant predictor of all population metrics in the regression analysis; this is because regression analysis primarily considers average difference.

Vegetation growth and habitat

The difference in tree species composition, tree mortality, post-fire basal area, and tree density provides evidence that Habitat 1 and Habitat 2 have distinct vegetation characteristics and consequently experienced different levels of fire intensity in 2013. This difference in fire intensity likely relates to vegetation density and fuel loading, because higher tree density is linked to higher fire intensity through fuel loading (Collins et al. 2019). This fire intensity is also evident through NDVI; the burned and unburned sites in Habitat 1 have much similar NDVIs than the burned and unburned sites in Habitat 2. Habitat 1 also displayed larger oscillations in NDVI, suggesting that the NDVI of the burned site in Habitat 1 is strongly affected by seasonal plants. These two habitats appear to have experienced different intensities of wildfire, which is evident in the current vegetation and habitat characteristics.

Vegetation growth emerged as a significant positive predictor for total abundance, total diversity, and tenebrionid abundance and diversity. The positive response in abundance and diversity to vegetation growth may result from post-fire soil being unfavorable for soil-dwelling decomposers until vegetation regrows (Gongalsky et al. 2012, Malmström 2010). When considering soil-dwelling assemblages, habitat destruction is an important factor that affects recovery after fire (Malmström 2010), and higher amounts of vegetation typically attract and support higher densities of herbivores (Malmström et al. 2009). Tenebrionids are primarily decomposers and rely on leaf litter for food, and tenebrionid abundance and diversity increases with higher herbaceous plant diversity (Liu et al. 2016). Carabid abundance and diversity was not linked to vegetation. There is not a significant relationship between carabid diversity and plant species diversity (Liu et al. 2016), suggesting that carabids are less linked to vegetation change.

Impact on beetle assemblages

Using two families of beetles as ecological indicators, it is clear that drought negatively affects soil-dwelling beetles in oak woodland ecosystems, and fire has positive impacts, regardless of habitat type or fire severity. This suggests that soil-dwelling beetles are impacted by long-term precipitation patterns, likely through changes in habitat characteristics (Chikoski et al. 2006). Fire promotes abundance and diversity independent of other factors, which suggests that the positive impacts of fire on habitat and nutrient availability are stronger than the direct mortality that can occur from high intensity wildfires (Gongalsky et al. 2012, Whelan 1995). Beetle abundance and diversity experiences larger oscillations in Habitat 1 than Habitat 2, suggesting that fire severity is linked to increased beetle movement between habitats. The Morgan fire occurred within a drought period and had positive impacts on diversity and abundance, suggesting that fire promotes abundance and diversity regardless of drought. Vegetation and habitat are important for decomposer organisms in the post-fire period, but not as important for predators. This is likely caused by the direct relationship decomposers have with vegetation (Gongalsky et al. 2012, Malmström 2010). Abundance was not affected by vegetation regrowth or habitat difference. Overall, drought has negative consequences for this ecosystem, fire promotes abundance and diversity, and the importance of vegetation growth depends on life history.

Limitations

Fully assessing the impacts of fire requires pre-fire and post-fire data. The sampling for this study started in July 2014, which was about a year after the burn occurred. Collecting pre-fire data, or beginning the data collection closer to when the burn happened, would have enabled a more complete illustration of the impacts of the burn. This was not feasible due to park permit restrictions. The aerial imagery available for the sampling dates from July 2014 to December 2015 did not correct for reflectance and had a resolution of 6.5 meters. This introduces more error into the imagery analysis, because it is not corrected for cloud reflectance, and the pixels used to compute the NDVI are larger. Additionally, the imagery for Mar 21, 2015 and Dec 11th, 2015 had 13% and 16% cloud cover respectively. This may have impacted the imagery analysis for those dates, as cloud cover can affect the computation of an NDVI by covering up vegetation. I was unable to calculate NDVI for Jan 12th, 2016 - July 9th, 2016, as well as March 22nd, 2015, and July 19th, 2015 because I could not access a computer to use the required software due to COVID-19. This was a limitation of the availability of high-quality imagery and computer lab availability; the NDVI analysis could be more accurate with more accurate and complete aerial imagery. More frequent vegetation surveys would have also added another element to my analysis, because vegetation species data would be available. More accurate imagery, pre-fire data, and more frequent vegetation surveys would have added value to my study.

Future directions

While I have shown significant impacts of drought, wildfire, and vegetation growth on two families of beetles, it necessitates further exploration into how fire intensity impacts beetle populations. I was able to graphically show differences in beetle population metrics over time, but more exploration is needed. This will be particularly relevant as we experience more large high intensity fires. Additionally, measuring soil differences before and after the fire would enable the estimation of fire severity, and thus understand the magnitude of the fire's effects on soil habitat. This difference cannot be captured by measuring fire intensity through tree mortality. Further exploration of the link between vegetation recovery and beetle populations would be valuable; what plants beetles rely on in a post-fire period is not fully understood, and NDVI is unable to capture that information.

Broader implications

California has been labeled as a climate change hotspot (Diffenbaugh et al. 2008). Thus, it is essential to understand how climate change will impact California ecosystems and the organisms that inhabit them. Fires are important for maintaining biodiversity, and droughts are detrimental to soil-dwelling beetles in oak woodland ecosystems. These findings have implications for future management plans for Mount Diablo and the San Francisco Bay Area. Management plans should consider introducing prescribed fire into the oak woodland ecosystems in the bay area to promote abundance and diversity of organisms. Future droughts will negatively impact these ecosystems but lowering diversity and abundance of organisms. Long term, these decreases in abundance and diversity may significantly impact biodiversity and ecosystem services. These implications of management practices and climate change on these ecosystems need to be considered when making forest management and climate policy decisions.

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REFERENCES

Andersen, A. N., and W. J. Muller. 2000. Arthropod responses to experimental fire regimes in an Australian tropical savannah: ordinal-level analysis. Austral Ecology 25:199-209.

ArcMap [GIS software]. 2019. Version 10.7. Redlands, CA: Environmental Systems Research Institute, Inc.

Beaudry, S., L. C. Duchesne, and B. CôTé. 1997. Short-term effects of three forestry practices on carabid assemblages in a jack pine forest. Canadian Journal of Forest Research 27:2065-2071.

Blanche, K. R., A. N. Andersen, and J. A. Ludwig. 2001. Rainfall-contingent detection of fire impacts: Responses of beetles to experimental fire regimes. Ecological Applications 11:86-96.

California Department of Resources.. 2020. California Data Exchange Center. URL http://cdec.water.ca.gov/index.html

Certini, G. 2005. Effects of fire on properties of forest soils: a review. Oecologia 143:1-10.

Chikoski, J. M., S. H. Ferguson, and L. Meyer. 2006. Effects of water addition on soil arthropods and soil characteristics in a precipitation-limited environment. Acta Oecologica-International Journal of Ecology 30:203-211.

Collins, B. M., J. D. Miller, E. E. Knapp, and D. B. Sapsis. 2019. A quantitative comparison of forest fires in central and northern California under early (1911-1924) and contemporary (2002-2015) fire suppression. International Journal of Wildland Fire 28:138-148.

Cuff, D., and E. Nardi. 2013. Mount Diablo fire revives debate over cattle grazing limits in state park.

Daly, A. J., J. M. Baetens, and B. De Baets. 2018. Ecological Diversity: Measuring the Unmeasurable. Mathematics 6:119.

Diffenbaugh, N. S., F. Giorgi, and J. S. Pal. 2008. Climate change hotspots in the United States. Geophysical Research Letters 35.

Evans, A. V., and J. N. Hogue. 2004. Introduction to California beetles. Berkeley: University of California Press, c2004, .

Flannigan, M. D., M. A. Krawchuk, W. J. de Groot, B. M. Wotton, and L. M. Gowman. 2009. Implications of changing climate for global wildland fire. International Journal of Wildland Fire 18:483-507.

Ghani, A., and S. Maalik. 2020. Assessment of diversity and relative abundance of insect fauna associated with Triticum aestivum from district Sialkot, Pakistan. Journal of King Saud University Science 32:986-995.

Gongalsky, K. B., A. Malmström, A. S. Zaitsev, S. V. Shakhab, J. Bengtsson, and T. Persson. 2012. Do burned areas recover from inside? An experiment with soil fauna in a heterogeneous landscape. Applied Soil Ecology 59:73-86.

Griffin, D., and K. J. Anchukaitis. 2014. How unusual is the 2012-2014 California drought? Geophysical Research Letters 41:9017-9023.

Hoekman, D., K. E. LeVan, G. E. Ball, R. A. Browne, R. L. Davidson, T. L. Erwin, C. B. Knisley, J. R. LaBonte, J. Lundgren, D. R. Maddison, W. Moore, J. Niemelae, K. A. Ober, D. L. Pearson, J. R. Spence, K. Will, and T. Work. 2017. Design for ground beetle abundance and diversity sampling within the National Ecological Observatory Network. Ecosphere 8:e01744.

Holliday, N. 1992. The Carabid Fauna (coleoptera, Carabidae) During Postfire Regeneration. Canadian Journal of Zoology-Revue Canadienne De Zoologie 70:440-452.

Intergovernmental Panel on Climate Change, (IPCC). Climate Change 2013: The Physical Science Basis. Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change. New York, NY, USA: Cambridge University Press; 2013.

Jansen, A. 1997. Terrestrial invertebrate community structure as an indicator of the success of a tropical rainforest restoration project. Restoration Ecology 5:115-124.

Johnston, J. M. 2000. The Contribution of Microarthropods to Aboveground Food Webs: A Review and Model of Belowground Transfer in a Coniferous Forest. The American Midland Naturalist 143:226-238.

Keeley, J. E. 2005. Fire history of the San Francisco East Bay region and implications for landscape patterns. International Journal of Wildland Fire 14:285-296.

Lencinas, M.V., F. J. Sola, J. Manuel Cellini, P. L. Peri, and G. Martinez Pastur. 2019. Land sharing in South Patagonia: Conservation of above-ground beetle diversity in forests and non-forest ecosystems. Science of the Total Environment 690:132-139.

Liu, J., F. Li, T. Sun, L. Ma, L. Liu, and K. Yang. 2016. Interactive effects of vegetation and soil determine the composition and diversity of carabid and tenebrionid functional groups in an arid ecosystem. Journal of Arid Environments 128:80-90.

Longcore, T. 2003. Terrestrial Arthropods as Indicators of Ecological Restoration Success in Coastal Sage Scrub (California, U.S.A.). Restoration Ecology 11:397-409.

Malmström, A. 2010. The importance of measuring fire severity—Evidence from microarthropod studies. Forest Ecology and Management 260:62-70.

Malmström, A., T. Persson, K. Ahlström, K. B. Gongalsky, and J. Bengtsson. 2009. Dynamics of soil meso- and macrofauna during a 5-year period after clear-cut burning in a boreal forest. Applied Soil Ecology 43:61-74.

Pearce, J. L., and L. A. Venier. 2006. The use of ground beetles (Coleoptera : Carabidae) and spiders (Araneae) as bioindicators of sustainable forest management: A review. Ecological Indicators 6:780-793.

Planet Team (2017). Planet Application Program Interface: In Space for Life on Earth. San Francisco, CA. https://api.planet.com.

Pressler, Y., J. C. Moore, and M. F. Cotrufo. 2019. Belowground community responses to fire: meta-analysis reveals contrasting responses of soil microorganisms and mesofauna. Oikos 128:309-327.

Restaino, J. C., and D. L. Peterson. 2013. Wildfire and fuel treatment effects on forest carbon dynamics in the western United States. Forest Ecology and Management 303:46-60.

RStudio Team. 2019. RStudio: Integrated Development for R. RStudio, Inc., Boston, MA URL http://www.rstudio.com/.

Samu, F., F. Kadar, G. Onodi, M. Kertesz, A. Sziranyi, E. Szita, K. Fetyko, D. Neidert, E. Botos, and V. Altbaecker. 2010. Differential ecological responses of two generalist arthropod groups, spiders and carabid beetles (Araneae, Carabidae), to the effects of wildfire. Community Ecology 11:129-139.

Seastedt, T. 1984. The Role of Microarthropods in Decomposition and Mineralization Processes. Annual Review of Entomology 29:25-46.

Van de Water, Kip M., and H. D. Safford. 2011. A Summary of Fire Frequency Estimates for California Vegetation Before Euro-American Settlement. Fire Ecology 7:26-58.

Weary, B. P., P. K. Mendez, S. E. Lew, B. L. Nyman, and K. W. Will. 2019. Performance of ramp and pitfall traps for arthropods and non-target vertebrates in Californian oak woodland and chaparral habitats. Pan-Pacific Entomologist 95:21-32.

Westerling, A. L. 2016. Increasing western US forest wildfire activity: sensitivity to changes in the timing of spring. Philosophical Transactions of the Royal Society B-Biological Sciences 371:20150178.

Whelan, R. J. 1995. The Ecology of Fire. Cambridge University Press.

Wikars, L. O. 1995. Clear-cutting before burning prevents establishment of the fire-adapted Agonum quadripunctatum (Coleoptera:Carabidae). Annales Zoologici Fennici 32:375-384.

Wikars, L. 1997. Effects of forest fire and the ecology of fire-adapted insects. Uppsala: Acta Universitatis Upsaliensis.

Williams, R. S., B. S. Marbert, M. C. Fisk, and P. J. Hanson. 2014. Ground-Dwelling Beetle Responses to Long-Term Precipitation Alterations in a Hardwood Forest. Southeastern Naturalist 13:138-155.

Yi, H., and A. Moldenke. 2005. Response of Ground-Dwelling Arthropods to Different Thinning Intensities in Young Douglas Fir Forests of Western Oregon. Environmental Entomology 34:1071-1080.