McCann *et al. Fire Ecology* (2025) 21:78
https://doi.org/10.1186/s42408-025-00428-6

Fire Ecology

Association for Fire Ecology

FIELD NOTE

Open Access

Postfire recovery trajectories of bulldozed versus burned chaparral eight years postfire

Erin McCann[1*], Zachary Flores[2], Janet Franklin[2,3], Marko J. Spasojevic[1,5], Nicole Molinari[4] and Loralee Larios[2,6]

Abstract

Background We examined vegetation diversity, structure, and composition on and off fuel breaks established during a 2013 wildfire in California chaparral shrublands. Vegetation was sampled 8 years following the fire to identify any persistent changes in structure or composition caused by this fire management activity, with implications for postfire vegetation recovery.

Results While species diversity and cover of lifeforms did not differ on and off fuel breaks, species composition and regeneration strategy of dominant shrubs differed significantly. Sites in fuel breaks were dominated by fast-growing subshrubs that regenerate from seeds and are more readily dispersed into sites—species that are typical indicators of the coastal sage scrub community. Sites off fuel breaks were characterized by a mix of resprouting and seeding shrubs typically associated with the chaparral community.

Conclusions Fuel breaks established by bulldozers during wildland firefighting have impacts on chaparral composition because the actions of the dozer remove soil seed banks and damage resprout "banks" (lignotubers). The permanence of these changes is likely to be related to the frequency and severity of fire suppression actions.

Keywords Disturbance intensity, Dispersal, Fire management, Vegetation recovery, Contingency lines

Resumen

Antecedentes Examinamos la diversidad, estructura y composición dentro y fuera de barreras de combustibles establecidas durante el incendio de vegetación en arbustales del chaparral de California en 2013. La vegetación fue muestreada 8 años después del incendio para identificar cualquier cambio persistente en la estructura y composición causada por esta actividad de manejo con sus implicancias para la recuperación de la vegetación post fuego.

Resultados Mientras que la diversidad de especies y la cobertura de las formas de vida no difirieron dentro y fuera de las barreras de combustibles, la composición de especies y las estrategias de regeneración de los arbustos dominantes variaron significativamente. Los sitios en las barreras de combustible estaban dominados por sub-arbustos de rápido crecimiento que regeneraron por semillas y son más rápidamente dispersados en los sitios especies que son indicadoras típicas de las comunidades del matorral costero–. Los sitios por fuera de las barreras de combustible estaban caracterizados por una mezcla de de arbustos rebrotantes y provenientes de semilla asociados típicamente con la comunidad del chaparral.

Conclusiones Las barreras de combustibles establecidas mediante bulldozers durante el combate de incendios de vegetación tienen impactos en la composición del chaparral dado que la acción de estos bolldozers remueve el

*Correspondence:
Erin McCann
emcca006@ucr.edu
Full list of author information is available at the end of the article

banco de semillas del suelo y daña los "bancos de rebrotes" (lignotúberes). La permanencia de estos cambios está probablemente relacionada con la frecuencia y la severidad de las actividades de supresión.

Introduction

With the increase in wildfire incidents in the Western USA (Balch et al. 2017; Gershunov et al. 2019; Safford et al. 2022), the scope of firefighting activities has similarly increased (Bayham et al. 2022). Fuel breaks are a common fire management tactic that serve a variety of goals from allowing firefighter access during an active wildfire to mediating fire spread and behavior (Agee et al. 2000; Hardy 2005). The term "fuel break" most commonly refers to areas of strategic value that are worthy of continued fuel reduction between fire events. Maintenance of fuel breaks occurs through a variety of methods and can result in permanent vegetation change that persists between fires. For permanent fuel breaks, differences in maintenance history (e.g., intervals between maintenance) as well as differences in how they were created (e.g., herbicide, mowing, mastication, targeted grazing, bulldozing) result in a gradient of vulnerability to future nonnative plant invasions as well as structural and compositional community-level changes (Merriam et al. 2006; Seipel et al. 2018; Grupenhoff and Molinari 2021). Invasions mediated by fuel breaks are of utmost concern as plant invasions may accelerate fire frequency and shifts from shrublands to nonnative grasslands (Syphard et al. 2019, 2022). Similar to fuel breaks, contingency lines (e.g., pre-attack lines) are areas where vegetation is removed for fire suppression operations, but this disturbance typically only occurs during fire events. Unlike permanent fuel breaks, vegetation on contingency lines is allowed to recover between fires, thereby presenting an opportunity to evaluate whether a less frequent management disturbance leads to reduced invasion potential. With more frequent fire in southern California, contingency lines created by bulldozers will become more common; therefore, understanding the recovery trajectory of chaparral (a dense evergreen shrubland) is essential to improving future management.

Vegetation recovery on fuel breaks created with bulldozers likely follows different pathways than the surrounding burnt landscape. Instead of having a thick layer of nutrient-rich ash (Neary et al. 1999), these areas are barren and often have had the topsoil and associated seedbank removed. They can experience soil compaction from bulldozer activities (Cullen et al. 1991) and may be the recipients of nonnative propagules carried on fire suppression equipment (Backer et al. 2004; Merriam et al. 2007). Within fire-prone ecosystems, many species have evolved regeneration strategies that allow them to rapidly colonize after a disturbance either through seedbanks or resprouting (Pausas and Keeley 2014). However, the scraping that occurs from bulldozers may remove soil seedbanks as well as damage or remove resprouting lignotubers (Green 1977), limiting vegetation recovery to dispersal-based species. When recovery is dependent on dispersal, the quality of the surrounding habitat plays a critical role in promoting either establishment by native species or nonnative invasions (Larios et al. 2013). Therefore, strategically placed fuel breaks known as contingency lines may not necessarily experience the negative vegetation shifts seen in more permanent breaks (Merriam et al. 2006; Seipel et al. 2018); however, few studies have investigated recovery in these areas.

Here we evaluate vegetation recovery on and off fuel breaks to better understand how the disturbance of contingency lines impacts the recovery of chaparral plant communities compared to the surrounding burnt vegetation. We sampled chaparral communities 8 years after wildfire to evaluate the postfire recovery trajectory of chaparral. Specifically, we asked how do the: (Q1) overall community composition, and cover, density, and height of different regeneration strategies, (Q2) diversity/richness of native and nonnative species, (Q3) cover of grasses/forbs/shrubs, and (Q4) cover of nonnative grasses and forbs differ in chaparral shrublands that burned compared to areas that experienced bulldozing. As fire suppression efforts grow, understanding the effects of suppression operations like fuel break creation on ecological communities will be key to developing mitigation measures and identifying restoration opportunities in recently burned landscapes.

Methods

Study site (Powerhouse Fire scar)

We evaluated fuel break recovery within the 2013 Powerhouse Wildfire Scar (Fig. 1). The Powerhouse Fire burned predominantly within the Angeles National Forest (ANF) in southern California for 9 days from May 30, 2013, through June 8, 2013, consuming 12,251 ha. The fire burned through 14 different vegetation communities that occurred over rugged terrain in the Sierra Pelona Mountains; however, the dominant burned vegetation type was chaparral shrubland, which made up ~86% of the burned area. Chaparral is a dense, evergreen, sclerophyllous, species-rich shrubland that is widespread in the Mediterranean climate of California (Keeley and Davis 2007).

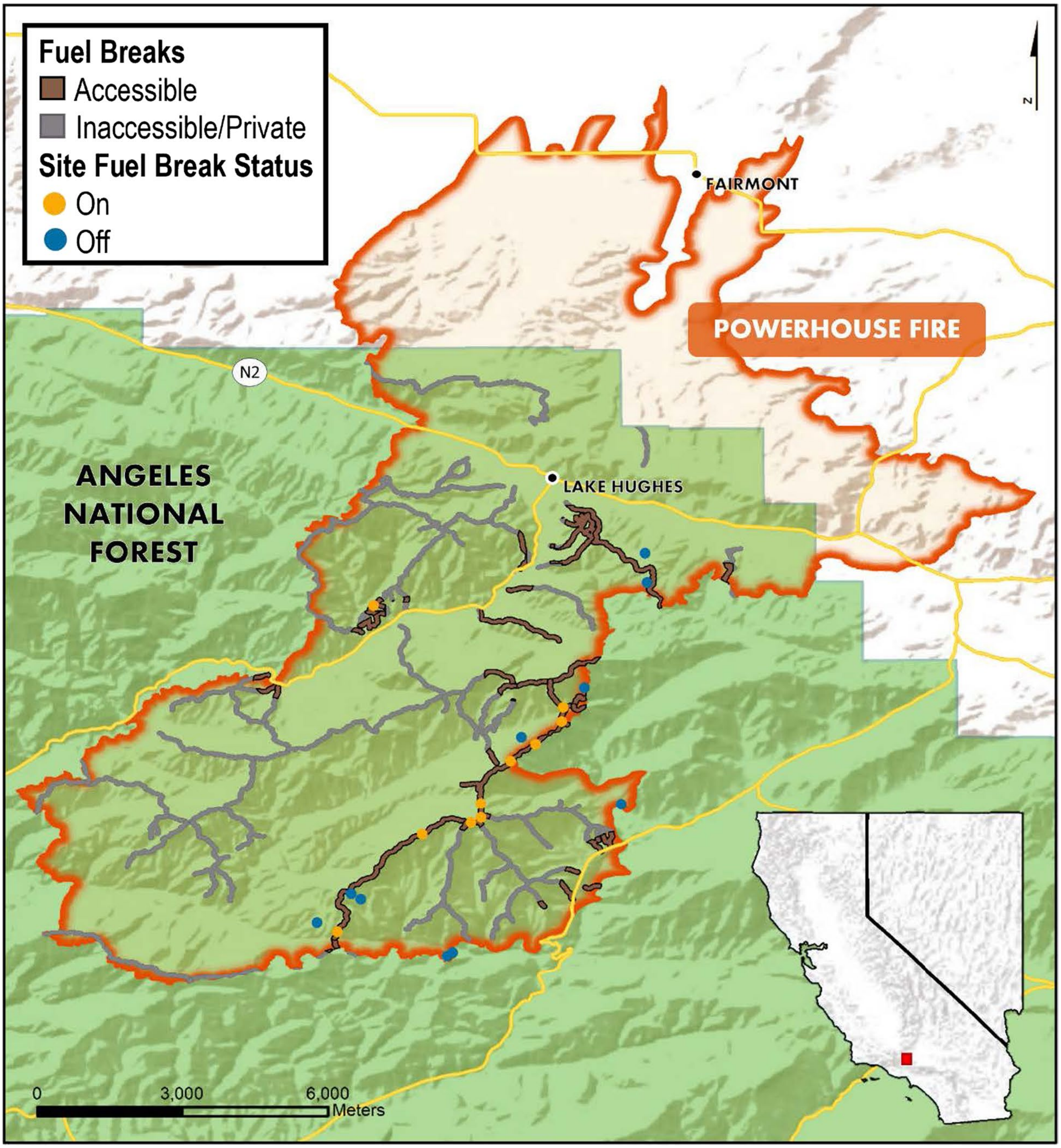

Fig. 1 Map of transect site locations throughout the Powerhouse Fire scar. All fuel breaks across the fire scar are represented on the map with the accessible fuel breaks in brown while the fuel breaks that were inaccessible and/or on privately owned land are depicted in gray. Orange circles indicate sites on fuel breaks while blue symbols represent the burned sites off the fuel breaks

Site selection

To select fuel break sites for this study, we obtained Geographical Information System (GIS) layers of fuel breaks, Powerhouse Fire scar, property ownership boundaries, and a 30-m resolution Digital Elevation Model (DEM). Using these layers, we randomly selected ten fuel break sites that were opened during the Powerhouse Fire. The maintenance history of our survey locations is unclear. Based on historical Google imagery, these sites experienced between 2 and 5 maintenance events between 1994 and 2013. This evaluation indicated maintenance consisted of either reopening with a dozer or cutting the brush via hand treatment, with dozer being the most common. The Land Management Plan on the Angeles National Forest recommends maintaining chaparral fuel breaks between 30 and 90 m, but the width of the fuel breaks in our study varied depending on the topography. We calculated the slope angle and aspect for each of the fuel break monitoring locations and used the same criteria to select sites with similar slope and aspect that burned within the Powerhouse Fire scar and did not experience disturbance from bulldozing.

Field sampling methods

Vegetation sampling was conducted from June through September 2019. We used the National Park Service vegetation monitoring scheme for the Mediterranean Coast Network (Tiszler et al. 2016) as a framework for our sampling design. This method consisted of three different sampling efforts to collect: (1) plant community composition (diversity, richness, abundance for all plants) via point-line intercept, (2) cover, (3) shrub density via a belt transect, and (4) shrub canopy height.

To assess community composition and cover within each site, we set up a 30-m transect. At 30-cm increments along the transect, we recorded every species that was hit by a 1-cm rod that was dropped vertically for a total of 100 sampling points per transect. We also recorded when multiple species of similar lifeforms (e.g., shrub, grass, forb) were intercepted at the same point (e.g., two shrubs one above the other). We were able to use these tallies of double counts to adjust our total estimate of lifeform cover so that we would not overestimate cover based on overlapping species.

Chaparral species generally exhibit one of three postfire regeneration strategies: (1) obligate resprouting (species that rely solely on resprouting from surviving tissue to regenerate postfire), (2) obligate seeding (species with fire-cued germination that only regenerate from seed postfire), or (3) facultative seeding (species that employ both strategies and either resprout or establish from seed). In the field, we categorized each individual shrub as either resprouting or recruiting from seed rather than assigning the shrub species to one of the three predefined regeneration strategies. This was particularly important for facultative seeders which are capable of both regeneration pathways. This distinction allowed us to determine whether different disturbance types (bulldozing vs only burning) would favor one regeneration pathway (resprout vs seeding) over the other.

For each site, species presence and abundance were used to calculate species richness and community diversity (Shannon-Wiener). We made these estimates for all species and computed them separately based on species origin (native, nonnative). We aggregated species cover into three groups based on lifeform—grass, forb, and shrub—and shrub regeneration type (seeder or resprouter) to describe general trends in recovery.

To estimate shrub density, we established a 1-m belt along the uphill side of the transect, for a total sampling area of a 1 m × 30 m rectangle. We recorded the total number of individuals for each shrub species and assigned them to one of four life stage/regeneration type categories (Fried et al. 2004) recruited from seed, resprouting, mature adult, dead. To identify resprouting individuals, we looked for a charred woody burl that would indicate the individual experienced fire and resprouted. Mature individuals were determined by the presence of reproductive structures (buds, flowers, fruit).

Finally, to estimate height, we took five measurements of maximum vegetative height for each species that was observed along the transect. We then averaged those values to get a species average height per transect. If less than five individuals per species were present in a transect, we sampled all individuals present.

Data analysis

First, a permutational ANOVA (PERMANOVA) was used to analyze the differences in species composition between the sites on and off the fuel breaks using the adonis function (999 permutations) in the vegan package (Oksanen et al. 2013). Differences in the communities were measured with Bray-Curtis dissimilarities calculated using the relative abundance of each species. Species relative abundance along each transect was computed using the decostand function in the vegan package (Oksanen et al. 2013), where the number of hits for each species was divided by the total number of hits across all species for a transect. We evaluated the homogeneity of group dispersion with permutational analysis of multivariate dispersion (PERMDISP) using the betadisper function. We then visualized the results using a nonmetric multidimensional scaling (NMDS; metaNMDS function). Finally, we calculated species scores with the envfit function to identify which species were driving differences in community patterns. Species that significantly contributed ($p < 0.05$) to differences between the communities were represented with arrows and labeled on the NMDS plot.

We also used a two-way ANOVA to evaluate differences in the relative cover and density of shrubs, with regeneration type (resprouter, seeder), fuel break status (off, on), and their interaction as factors. To calculate regeneration relative cover, we summed all the hits for all species based on their regeneration type and divided it by the total number of hits across all species on a transect and multiplied by 100. To examine the differences in height between regeneration types on and off fuel breaks,

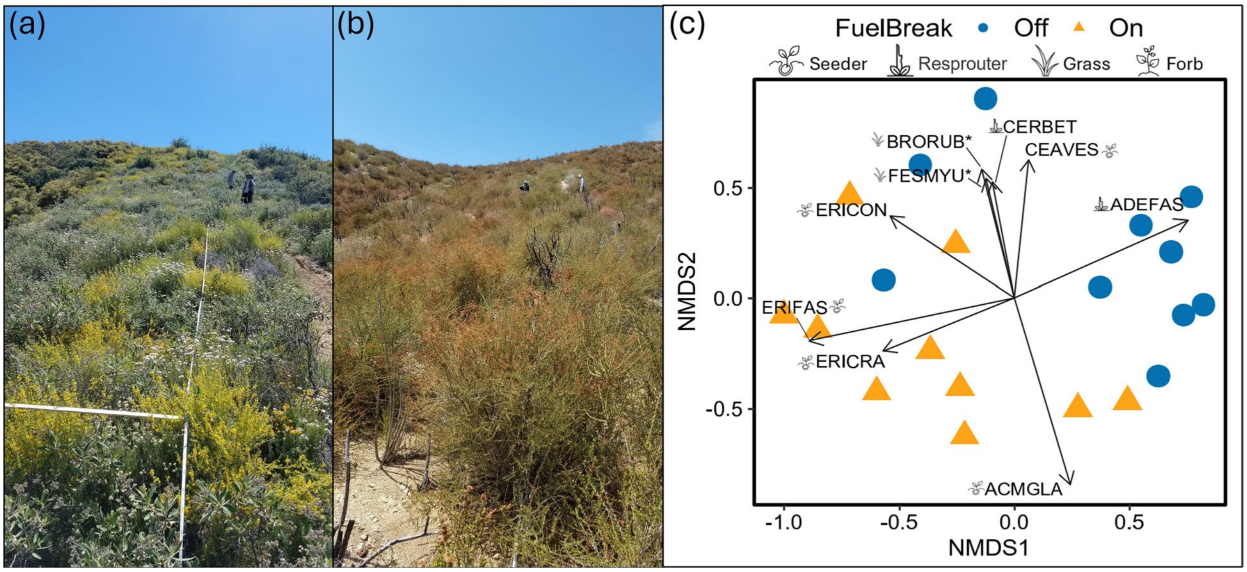

Fig. 2 Species composition between sites on and off the fuel breaks. **a** Photo of species composition on a fuel break highlighting the three species ACMGLA, ERIFAS, and ERICRA that are associated with fuel breaks. **b** Photo of species composition off a fuel break highlighting high abundance of ADEFAS. **c** Nonmetric multidimensional scaling (NMDS, stress value = 0.16) diagram showing the dissimilarity (Bray-Curtis index) of the species composition between the sites on (orange, triangles) and off (blue, circles) the fuel breaks. Each point represents a survey transect. The arrows represent the contribution of each species on each axis of the NMDS, and the compositional dissimilarity is indicated by the spread along the two NMDS axes. Species are coded by their life form and shrubs are separated by their regeneration type. Names for species codes can be found in Table S1 and model summary statistics in Table S2. Photos courtsey of Sameer S. Saroa

we calculated the average height for each shrub species observed within a transect. We used a mixed effects model for evaluating differences in height, with regeneration type (resprouter, seeder) and fuel break status (off, on) and their interaction as fixed factors and species nested with regeneration type as a random effect.

To evaluate differences in diversity and richness, we used two-sample *t*-tests to compare total, native, and nonnative richness and diversity between the sites on and off the fuel breaks. Next, to test for differences in relative cover of the shrubs, forbs, and grasses, we used a two-way ANOVA with lifeform (grasses, forbs, shrubs) and fuel break status (off, on) and their interaction as factors. We calculated relative cover as total hits for a life form divided by the unique total life form hits × 100, cover adjusting for double hits of a similar life form. The total hits for a life form were calculated as the total hits for a life form minus double hits for that life form (e.g., where two overlapping shrubs were observed as the same point). The unique total life form hits were calculated as the unique total life form hits (e.g., all grass, forb, shrub hits) minus the total number of double hits across all life forms. Additionally, for the forbs, a two-way ANOVA with fuel break status, native status (native, nonnative), and their interaction as factors was used to evaluate differences in the relative cover of nonnatives across the two site types. For grasses, native species were not present consistently or at high enough abundances to compare to nonnative species; therefore, we used a two-sample *t*-test to determine the difference in relative cover of nonnative grasses between sites on and off the fuel breaks. We used natural log transformations to meet assumptions of normality for regeneration density, height, and life form cover analyses, where we initially assessed normality via a Shapiro-Wilk test on the residuals of the models and visually with a Q-Q plot for ANOVA and mixed effects models and a Shapiro-Wilk test on the response variable for *t*-tests. Tukey HSD post hoc tests were applied to test any significant interactions. All analyses were performed in the R software environment (version 4.2.2, R Core Team, 2019).

Results

Species composition

The species composition of the transects on fuel breaks significantly differed from the transects off fuel breaks ($R^2 = 0.20$, $F = 4.56$, stress = 0.16, $p = 0.007$; Fig. 2, Table S2). However, there were no significant differences in dispersion between the two groups ($p = 0.93$). The fuel break sites were largely characterized by the presence of two native coastal sage scrub shrub species: *Acmispon glaber* (Fabaceae) and *Eriogonum fasciculatum* (Polygonaceae) (Sawyer & Keeler-Wolf 1995) and one chaparral species: *Eriodictyon crassifolium* (Namaceae)

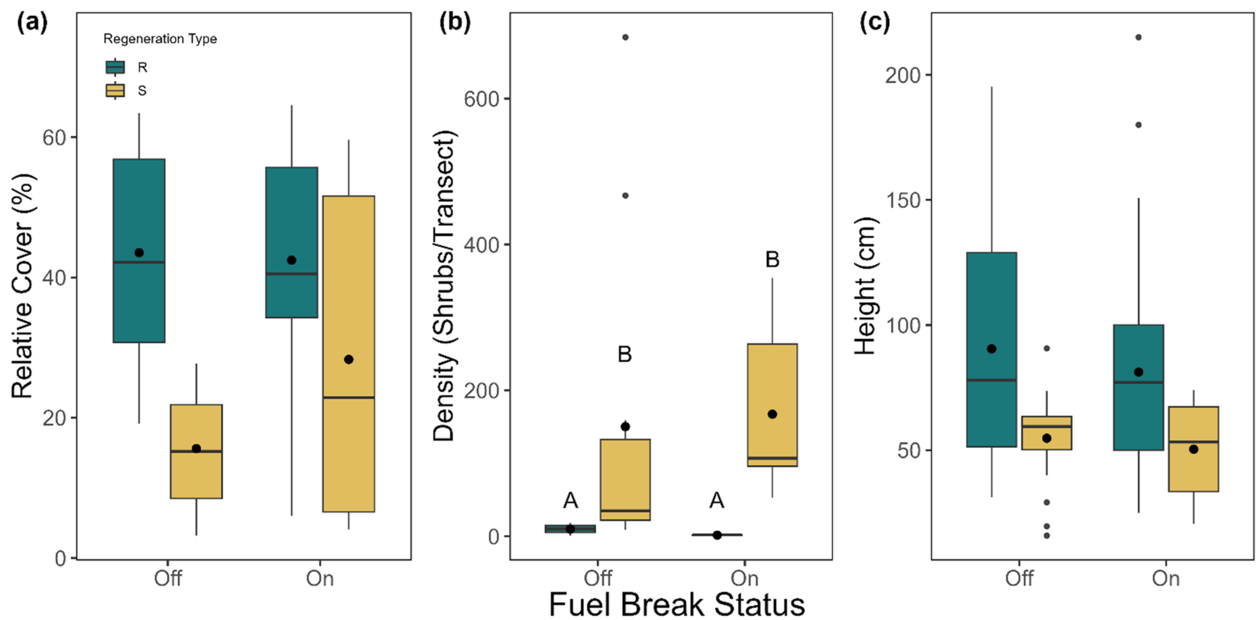

Fig. 3 Shrub relative cover (**a**), density (**b**), and height (**c**) based on the method of recolonization, on and off fuel breaks, 8 years postfire. Shrubs were categorized based on whether they regenerated by resprouting post-distrubance ("R") and "S" representing individuals that germinated from seed post-distrubance. Connecting letters indicate post hoc significant differences at $p < 0.05$ for a significant fuel break and regeneration type interaction. Full summary statistics are available in Table S3

(Montalvo et al. 2017). Burned sites were characterized by the presence of the native chaparral shrubs: *Adenostoma fasciculatum* (Rosaceae), *Ceanothus vestitus* (Rhamnaceae), and *Cercocarpus betuloides* (Rosaceae) and the nonnative grasses: *Bromus rubens* (Poaceae) and *Festuca myuros* (Poaceae).

Regeneration types

The relative cover of individuals regenerating from seed vs those that resprouted following the fire did not significantly differ between the sites on and off the fuel breaks (regeneration type: $p = 0.081$; regeneration × fuel break: $p = 0.22$; Table S3). For both the fuel breaks (42.46% ± 17.70) and the burned sites (43.53% ± 17.77), the relative cover of resprouters was very similar (Fig. 3a). While the fuel breaks had nearly double the average relative cover of seeders (28.31% ± 23.26) compared to the burned sites (15.60 ± 8.45), there was high variability across sites, with relative cover of seeders ranging from 4 to 59%. Unlike relative cover, the density of seeders and resprouters differed between the two site types (regeneration type: $p < 0.0001$; regeneration × fuel break: $p = 0.007$). However, these differences were driven not by disparities in seeder (post hoc, $p = 0.23$) or resprouter (post hoc, $p = 0.16$) density between the two sites, but by both site types having a significantly higher density of seeders compared to resprouters (post hoc, fuel break-burned: $p < 0.0001$, burned-fuel break: $p = 0.0001$). On average, the fuel breaks had nearly 100 times more seeders (167 shrubs/transect) than resprouters (1.67 shrubs/transect). Burned sites showed less of a disparity with 15 times more seeders (150 shrubs/transect) than resprouters (10 shrubs/transect). Shrubs did not differ in height between the fuel breaks or regeneration types (regeneration type: $p = 0.073$, regeneration × fuel break: $p = 0.239$; Table S4) with mean height ranging from 50.37 ± 19.23 cm for seeders on fuel breaks to a high of 90.45 ± 47.32 cm for resprouters off fuel breaks (Fig. 3c, Table S4).

Diversity & richness

Total ($p = 0.51$), native ($p = 0.70$), and nonnative ($p = 0.94$) species richness did not significantly differ on and off fuel breaks (Table S5), although there tended to be on average about three times more native species compared to nonnative species across all sampling sites independent of fuel break status. Furthermore, there was no difference in native ($p = 0.51$) or nonnative diversity ($p = 0.54$) on and off fuel breaks, and native diversity tended to be 2–3 times greater than nonnative diversity across all sites regardless of fuel break status.

Relative cover of grasses/forbs/shrubs

The relative cover of the three lifeforms differed from each other ($p < 0.0001$), but this was not influenced by fuel break status (fuel break: $p = 0.121$; fuel break × life form: $p = 0.322$; Fig. 4, Table S3). For both site types, shrubs

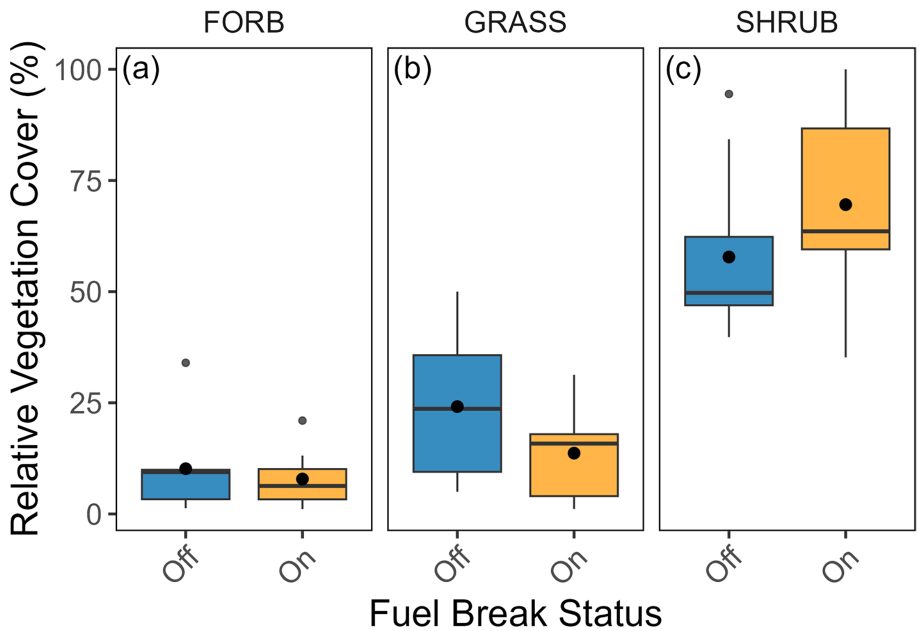

Fig. 4 Comparison of relative vegetation cover of **a** forbs, **b** grass, and **c** shrubs between transects off (blue) and on (yellow) the fuel breaks. The bolded horizontal lines represent data medians; the solid dots represent means. Relative cover of forb, grass, and shrub cover pooled across fuel break were all significantly different from each other, but there was no effect of fuel break. Full summary statistics are available in Table S3

were the dominant life form, making up 69.57% ± 19.79 (mean ± 1SD) of the relative cover on the fuel breaks transects and 57.79% ± 18.19 of the transects off the fuel break. Overall, the relative cover of shrubs was significantly greater than both grasses (post hoc, $p < 0.0001$) and forbs (post hoc, $p < 0.0001$). Grasses were the second most dominant lifeform, with significantly greater relative cover than forbs (post hoc, $p = 0.047$) and a mean relative cover of 14.30% ± 10.70 on the fuel breaks and 24.15% ± 17.41 off the fuel breaks. Forbs were the least prevalent both on (7.87% ± 6.67) and off (10.18% ± 11.14) the fuel breaks. Of the three functional groups, only forbs and grasses had native and nonnative species present both on and off the fuel breaks. For the forbs, the relative cover of native and nonnative species did not significantly differ from each other ($p = 0.596$; Table S3), and no difference existed relative to site type (fuel break: $p = 0.965$; fuel break × species origin: $p = 0.902$). As for grasses, native grasses were only present on burned sites, so only nonnative grass cover was compared across sites, and we found no significant difference in nonnative grass cover on (13.70% ± 10.35) and off the fuel breaks (23.52 ± 16.70, $t(9.77) = -1.35$; $p = 0.21$).

Discussion

Overall, our examination of the community structure, diversity, and composition in burned and contingency line sites demonstrates that the disturbance dynamics associated with bulldozing can alter post-disturbance community recovery trajectories but not necessarily promote plant invasions. Specifically, differences in community recovery emerged predominantly as differences in community composition, while community diversity and structure were largely similar on and off the fuel breaks. We found key differences in community composition on and off the fuel breaks (Q1). These compositional differences were most notably driven by a shift in the abundance of coastal sage scrub alliance species that more readily recruit from seed in the fuel breaks, while the burned sites were primarily characterized by species associated with the chaparral community. We found minimal differences in the recolonization strategy of shrubs off and on the fuel breaks. This was primarily due to the dynamic response of seedling recruitment across the study sites (e.g., highly variable cover on fuel breaks, a couple of high-density sites). In terms of community diversity (Q2), we found comparable native diversity and richness, as well as nonnative diversity and richness, both on and off the fuel breaks. Furthermore, the relative cover (Q3) of all three vegetation types and the cover of native and nonnative forbs (Q4) remained similar on and off the fuel breaks, suggesting that community recovery was not considerably impacted by the type of disturbance (bulldozing vs burning). These differences in community composition and regeneration types within these contingency line sites illustrate that intermittent fuel breaks may support native communities, albeit with different species, but with similar structural function as the surrounding sites within the fire scar.

The differences in composition between the two site types highlight how fuel breaks promote different patterns of colonization and recovery. Unlike the burned sites whose plant composition comprised a diverse assemblage of species including resprouting shrubs, the fuel break sites were predominantly structured by seeders. This pattern suggests that composition in the fuel breaks was shaped by species capable of dispersing into the fuel breaks. For example, a key indicator species on fuel breaks *Eriogonum fasciculatum* establishes postfire by dispersal (Minnich and Dezzani 1998). This reliance on colonization through dispersal is likely a result of the mechanisms used to establish fuel breaks. Fuel breaks are established to reduce fuel continuity by removing vegetation and scraping areas free of organic soil layers to expose the bare mineral soil beneath (Keeley 2002; Merriam et al. 2006; Shinneman et al. 2019). This process, while it can be effective in managing fire spread by facilitating firefighter access (Syphard et al. 2011), can severely disturb sites as it can cause soil compaction as well as the removal of the seedbank and even basal buds (Busse et al. 2014). These compositional differences between the fuel break and burned sites suggest that proximity to intact seed sources will be key to the recovery of these areas and may mediate disparities in long-term recovery instigated by divergent patterns in initial colonization.

Reliance on dispersal heightens the vulnerability of these sites as they are not as quickly colonized post-disturbance, leaving them susceptible to invasion. While in our study the community in the contingency lines shifted toward the more disturbance-tolerant native vegetation of the coastal sage scrub community, there is potential for more drastic degradation to occur. A long-term study near the Powerhouse Fire found that over a 79-year period, 75% of coastal sage scrub plots converted to annual grass, while only 1/3 of chaparral converted. These findings suggest that sage scrub may be more susceptible to invasion and less stable on the landscape compared to chaparral (Dewees et al. 2022). Other studies evaluating differences in community composition on recovering fuel breaks have found significantly higher levels of nonnative species abundance and cover in fuel breaks relative to burned sites (Merriam et al. 2006). The lack of nonnative species present in the contingency lines in the Powerhouse Fire scar can be attributed to an overall lack of nonnative species present in most areas we sampled. Other studies that did find abundant nonnative species in fuel breaks generally found that these sites either had high abundances of nonnatives pre-fire or were near roads and highways which can dramatically increase the potential for nonnative introduction (Merriam et al. 2006; Weinberger & Kaczynski 2022). Because fuel break establishment provides such an opportunity for nonnative introduction, as fire dynamics shift due to climate change, enhanced human ignitions, and other factors, the potential for these more dramatic community shifts is heightened (Cox et al. 2014; Syphard et al. 2022).

Within California, as fire activity continues to increase and fire and climate interactions continue to unfold (Keeley and Syphard 2016; Williams et al. 2019), we can expect that contingency lines will need to be opened more frequently. In fact, a few of the ridgelines surveyed in our analysis were bulldozed again during the 2020 Lake Fire. Increased frequency of fuel break maintenance has been shown to be significantly related to nonnative species cover (Merriam et al. 2006). Therefore, as contingency lines are shifted toward more permanent features on the landscape in response to increases in fire frequency, nonnative invasion may likely increase in regularity and severity. Furthermore, nonnative seed sources may expand into adjacent burned sites and disrupt native recovery (Merriam et al. 2007). While the fuel breaks in the Powerhouse Fire did not show signs of nonnative invasion, shifts to more permanently maintained fuel breaks will require more monitoring to minimize future degradation.

With more frequent disturbance to contingency lines, it is critical to consider potential mitigation measures that can be taken to minimize repeat short interval disturbances and the introduction of nonnatives. Firstly, survey efforts centered on early detection of nonnatives and eradication of small infestation of weeds are critical (Floyd et al. 2006). In addition to these monitoring efforts, various management practices can mitigate the potential for degradation of communities within contingency lines. For instance, rather than continually reopening the same areas, it may be beneficial to opt for opening alternative nearby areas to allow for longer periods of recovery between reopening. Additionally, cleaning of equipment used to create and maintain fuel breaks could significantly reduce the potential for the introduction of nonnative species (Tu et al. 2001). Finally, once the fire is contained, efforts to return burned soil and excavated vegetation to the opened area are important for restoring the native seedbank, creating microsites, and deterring unwanted use (e.g., off-highway vehicles, user-created trails) that could impede vegetation recovery.

In summary, a key result from this work is that disturbance by bulldozers altered the recovery trajectory of these sites. Specifically, fire-adapted species associated with the chaparral community were no longer present, and these bulldozed areas were dominated by more disturbance-tolerant coastal sage scrub subshrubs that often colonize via seed. Since our surveys in 2018, this site has experienced more fire. The Lake Fire in 2020 resulted in the reopening of some of the fuel breaks studied here. As fires continue to become more frequent in southern California, it will be important to invest in monitoring and evaluation tools to protect these communities from invasion and guide postfire restoration.

Supplementary Information

The online version contains supplementary material available at https://doi.org/10.1186/s42408-025-00428-6.

Supplementary Material 1.

Acknowledgements
We thank Sameer Sunny Saroa, Michelle Ortiz, and Jessica Du for conducting data collection and the Angeles National Forest for field access.

Authors' contributions
LL and NM conceived of the project idea. EM conducted site selection; ZF collected and entered data. EM, ZF, and LL conducted the analyses, and EM and LL wrote the first draft of the manuscript. JF, MJS, and NM provided feedback on the study design, analysis, and manuscript drafts. All authors read and approved the final manuscript.

Funding
This work was supported by the United States Department of Agriculture Forest Service.

Data availability
The datasets generated and/or analyzed during the current study are available in the Dryad data repository upon acceptance.

Declarations

Ethics approval and consent to participate
Not applicable.

Consent for publication
Not applicable.

Competing interests
The authors declare that they have no competing interests.

Author details
[1]Department of Evolution, Ecology and Organismal Biology, University of California Riverside, Riverside, CA 92521, USA. [2]Department of Botany and Plant Sciences, University of California Riverside, Riverside, CA 92521, USA. [3]Center for Open Geographical Sciences, Department of Geography, San Diego State University, 5500 Campanile Drive, San Diego, CA 92182, USA. [4]USDA Forest Service, Pacific Southwest Region, Regional Ecology Program, 1980 Old Mission Drive, Solvang, CA 93463, USA. [5]Environmental Dynamics and Geo-Ecology Institute, University of California Riverside, Riverside, CA 92521, USA. [6]Center for Conservation Biology, University of California Riverside, 1435 Boyce Hall, Riverside, CA 92521, USA.

Received: 5 March 2025 Accepted: 15 October 2025
Published online: 28 November 2025

References
Agee, J.K., B. Bahro, M.A. Finney, P.N. Omi, D.B. Sapsis, C.N. Skinner, J.W. van Wagtendonk, and C.P. Weatherspoon. 2000. The use of shaded fuelbreaks in landscape fire management. *Forest Ecology and Management* 127 (1–3): 55–66.

Backer, D.M., S.E. Jensen, and G.R. McPherson. 2004. Impacts of fire-suppression activities on natural communities. *Conservation Biology* 18: 937–946.

Bayham, J., J.K. Yoder, P.A. Champ, and D.E. Calkin. 2022. The economics of wildfire in the United States. *Annual Review of Resource Economics* 14 (1): 379–401.

Busse, M.D., K.R. Hubbert, and E.E.Y. Moghaddas. 2014. *Fuel reduction practices and their effects on soil quality*. U.S. Forest Service, Department of Agriculture. General Technical Report PSW-GTR-241, 156. California: Pacific Southwest Research Station. Albany.

Cox, R.D., K.L. Preston, R.F. Johnson, R.A. Minnich, and E.B. Allen. 2014. Influence of landscape-scale variables on vegetation conversion to exotic annual grassland in southern California, USA. *Global Ecology and Conservation* 2: 190–203.

Cullen, S.J., C. Montagne, and H. Ferguson. 1991. Timber harvest trafficking and soil compaction in western Montana. *Soil Science Society of America Journal* 55: 1416–1421.

Dewees, S. L., C. M. D'Antonio, and N. Molinari. 2022. Determining potential drivers of vegetation change in a Mediterranean environment. *Ecosphere* 13 (12) : e4313.

Floyd, M.L., D. Hanna, W.H. Romme, and T.E. Crews. 2006. Predicting and mitigating weed invasions to restore natural post-fire succession in Mesa Verde National Park, Colorado, USA. *International Journal of Wildland Fire* 15: 247–259.

Fried, J.S., C.L. Bolsinger, and D. Beardsley. 2004. *Chaparral in southern and central coastal California in the mid-1990s: area, ownership, condition, and change*. U.S. Forest Service, Department of Agriculture. Resource Bulletin PNW-RB-240, 86. Oregon: Pacific Northwest Research Station. Portland.

Gershunov, A., J. Guzman-Morales, D.A. Bishop, J.K. Balch, and D.P. Lettenmaier. 2019. Observed impacts of anthropogenic climate change on wildfire in California. *Earth's Future* 7 (8): 892–910.

Green L.R. 1977. *Fuelbreaks and other fuel modification for wildland fire control*. US Forest Service, Department of Agriculture. Agricultural Handbook 499, 79. Washington, DC.

Grupenhoff, A., and N. Molinari. 2021. Plant community response to fuel break construction and goat grazing in a Southern California shrubland. *Fire Ecology.* 17 (1): 1–11.

Hardy, C.C. 2005. Wildland fire hazard and risk: Problems, definitions, and context. *Forest Ecology and Management* 211: 73–82.

J.K. Balch, B.A. Bradley, J.T. Abatzoglou, R.C. Nagy, E.J. Fusco, & A.L. 2017. Mahood, Human-started wildfires expand the fire niche across the United States. *Proceedings of the National Academy of Sciences of the United States of America* 114 (11) 2946–2951. https://doi.org/10.1073/pnas.1617394114

Keeley, J.E. 2002. Fire management of California shrubland landscapes. *Environmental Management* 29: 395–408.

Keeley, J.E., and A.D. Syphard. 2016. Climate change and future fire regimes: Examples from California. *Geosciences* 6 (3): 37.

Keeley, J. E., & F. W. Davis. 2007. Chaparral. In Terrestrial vegetation of California, ed. M. G. Barbour, T. Keeler-Wolf, & A. A. Schoenherr, 339–366. Oakland: University of California Press.

Larios, L., R.J. Aicher, and K.N. Suding. 2013. Effect of propagule pressure on recovery of a California grassland after an extreme disturbance. *Journal of Vegetation Science* 24: 1043–1052.

Merriam, K.E., J.E. Keeley, and J.L. Beyers. 2006. Fuel breaks affect nonnative species abundance in Californian plant communities. *Ecological Applications* 16 (2): 515–527.

Merriam, K.E., J.E. Keeley, and J.L. Beyers. 2007. *The role of fuel breaks in the invasion of nonnative plants*. U.S. Geological Survey, Department of the Interior. Scientific Investigations Report 2006–5185, 69. Reston, Virginia.

Minnich, R. A., and R. J. Dezzani. 1998. Historical decline of coastal sage scrub in the Riverside-Perris plain, California. *Western Birds* 29:366–391.

Montalvo, A. M., E. C. Riordan, and J. Beyers. 2017. *Plant profile for Eriodictyon crassifolium*. U.S. Forest Service, Department of Agriculture & Riverside-Corona Resource Conservation District. Native plant recommendations for Southern California ecoregions. California: Pacific Southwest Research Station, Riverside.

Neary, D.G., C.C. Klopatek, L.F. DeBano, and P.F. Ffolliott. 1999. Fire effects on belowground sustainability: A review and synthesis. *Forest Ecology and Management* 122 (1–2): 51–71.

Oksanen, J., Blanchet, F. G., Kindt, R., Legendre, P., Minchin, P. R., O'Hara, R. B., Simpson, G. L., Solymos, P., Henry, M., Stevens, H., & Wagner, H. 2013. Vegan: community ecology package. R package version 2.6–4. https://vegandevs.github.io/vegan/

Pausas, J.G., and J.E. Keeley. 2014. Evolutionary ecology of resprouting and seeding in fire-prone ecosystems. *New Phytologist* 204 (1): 55–65.
Safford, H.D., A.K. Paulson, Z.L. Steel, D.J.N. Young, and R.B. Wayman. 2022. The 2020 California fire season: A year like no other, a return to the past, or a harbinger of the future? *Global Ecology and Biogeography* 31 (10): 2005–2025.
Sawyer, John O.., and Todd Keeler-Wolf. 1995. *A manual of California vegetation*. Sacramento: California Native Plant Society.
Seipel, T., L.J. Rew, K.T. Taylor, B.D. Maxwell, and E.A. Lehnhoff. 2018. Disturbance type influences plant community resilience and resistance to *Bromus tectorum* invasion in the sagebrush steppe. *Applied Vegetation Science* 21 (3): 385–394.
Shinneman, D.J., M.J. Germino, D.S. Pilliod, C.L. Aldridge, N.M. Vaillant, and P.S. Coates. 2019. The ecological uncertainty of wildfire fuel breaks: Examples from the sagebrush steppe. *Frontiers in Ecology and the Environment* 17: 279–288.
Syphard, A.D., J.E. Keeley, and T.J. Brennan. 2011. Comparing the role of fuel breaks across southern California national forests. *Forest Ecology and Management* 261 (11): 2038–2048.
Syphard, A.D., T.J. Brennan, and J.E. Keeley. 2019. Drivers of chaparral type conversion to herbaceous vegetation in coastal southern California. *Diversity and Distributions* 25 (1): 90–101.
Syphard, A.D., T.J. Brennan, H. Rustigian-Romsos, and J.E. Keeley. 2022. Fire-driven vegetation type conversion in Southern California. *Ecological Applications* 32 (6): e2626.
Tiszler, J., D. Rodriguez, K. Lombardo, T. Sagar, L. Aguilar, L. Lee, T. Handley, K. McEachern, L. Starcevich, M. Witter, T. Philippi, and S. Ostermann-Kelm. 2016. *Terrestrial vegetation monitoring protocol for the Mediterranean Coast Network—Cabrillo National Monument, Channel Islands National Park, and Santa Monica Mountains National Recreation Area: narrative, version 1.0.* U.S. National Parks Service. Natural Resource Report 2016-1296. Fort Collins, Colorado.
Tu, M., C. Hurd, J.M. Randall, and The Nature Conservancy. 2001. *Weed control methods handbook: tools & techniques for use in natural areas*. U.S. Government Documents. Report 533. Utah.
Weinberger, H.S., and K.M. Kaczynski. 2022. Dozer lines influence post-fire plant community recovery. *Madroño* 69: 74–87.
Williams, A. P., JT Abatzoglou, J. T., Gershunov, A., Guzman-Morales, J., Bishop, D. A., Balch, J. K., & Lettenmaier, D. P. 2019. Observed impacts of anthropogenic climate change on wildfire in California. *Earth's Future* 7(8): 892–910.

Publisher's Note