The impact of fire on the reported presence of animals in California Deserts using open-source data

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Abstract

Changing fire regimes across southwest North American deserts may impact endangered animal communities endemic to the region. This study examines the impact of fires on the occurrence of endangered animal species (ES) in California desert systems and evaluates ES recovery trends using open-source data—mostly collected through citizen science—retrieved from the Global Biodiversity Information Facility. Mean annual NDVI was used to evaluate vegetation productivity in fire impacted desert regions. ES occurrence records were fit to generalized linear mixed models and compared pre- and post-fire to evaluate ES response to fire disturbance. ES recovery was evaluated using a incidence-based ChaoSørensen similarity index. Burned regions had higher vegetation productivity than unburned regions in some, but not all, deserts. ES continue to visit burned habitat, even 19 years after a fire. Findings suggest ES resiliency to fire disturbance, likely through habitat-use modification, and support implementing citizen science data in future ecosystem monitoring.

Keywords

Desert, fire, GBIF, NDVI, citizen science, grass/fire cycle, endangered species, open-source data, disturbance, recovery

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List of Abbreviated Terms and Acronyms

BACI   Before/after control/impact framework

ES   Endangered animal species

GLM   Generalized linear model

GLMM   Generalized linear mixed model

GBIF   Global Biodiversity Information Facility

MOJ   Mojave Desert

NDVI   Normalized Difference Vegetation Index

NIR   Reflected near-infrared light

R   Reflected red light

SJD   San Joaquin Desert

SON   Sonoran Desert

Introduction

Ecological background

Historically, the landscape and climate of California's desert regions has not supported large and frequent wildfires (Humphrey, 1974; Brown & Minnich, 1986; Syphard & Keeley, 2020; Park et al., 2021). Elevation and geography influence vegetation type in the American southwest desert regions, which, in turn, influences fire frequency and size (Brooks & Matchett, 2006; Brooks et al., 2018; Sugihara et al., 2018; McAuliffe, 2020). Much of desert systems in California consist of sparse vegetation creating patchy, low surface fuels that hampers fires spreading far from the ignition point (Brooks et al., 2018; Wills, 2018). Although lightning frequency–the main ignition source for most desert wildfires–is higher in deserts than in any other California bioregion (van Wagtendonk & Cayan, 2008), low rainfall reduces availability of fine fuel and, therefore, reduces the chances of fire spreading from ignition and creates a long fire-return interval (Brown & Minnich, 1986; Brooks & Matchett, 2006; Minnich, 2018). Nevertheless, variation in precipitation and vegetative composition throughout the southwest desert systems creates a geographically varied fire regime that in turn is likely to influence patterns of regional biodiversity (Pastro et al., 2011; Diffendorfer et al., 2012; Dorph et al., 2020).

Ecosystems worldwide are experiencing rapid ecological change (Vitousek, 1994; Chen et al., 2011; Poloczanska et al., 2013; Lenoir & Svenning, 2015; Pecl et al., 2017; Smale et al., 2019) and the effects on both ecosystem function and biodiversity are large-scale and varying (Chen et al., 2011; Luque et al., 2013; Pecl et al., 2017). Throughout the 20th century, deserts in California have experienced episodes of drought punctuated by wetter periods–usually following a 30-year cyclical trend (Hereford et al., 2006). Most recently, beginning around 1999, the southwestern North American deserts have been under extreme dry conditions–interrupted

only briefly in 2005, 2011, and 2019 (Cook et al., 2004, 2010; Mann & Gleick, 2015; Woodhouse et al., 2010). Although paleoclimatic records indicate that the most recent drought is not the most severe, warming temperatures and increased human activities (e.g., conversion of desert land to farmland, fire suppression, urbanization, and introduction of invasive species) will exacerbate the impacts of future droughts in California ecosystems (Woodhouse et al., 2010; Diffenbaugh et al., 2015). Climate models for the mid-21st century project increased fire activity in California's desert lands as a response to increased aridity and drought frequency (Abatzoglou & Kolden, 2011; Krawchuk & Moritz, 2012). Non-native annual grasses, such as *Bromus* and *Schismus* spp, can withstand prolonged drought conditions and outcompete native plants during wetter years (Balch et al., 2013; Horn & St. Clair, 2017; Moloney et al., 2019). The dry/wet cycle combined with the invasion of non-native annual grasses has led to previously barren space to be filled with continuous fine fuel and, thus, further increase the occurrence and spread of wildfires (Brooks et al., 2004; Balch et al., 2013; Fusco et al., 2019). Moreover, fires can reduce native vegetation and further promote the spread of invasive plant species creating what is termed an invasive grass/fire cycle (Brown & Minnich, 1986; D'Antonio & Vitousek, 1992; Brooks et al., 2004; Brooks & Zouhar, 2008; Esque et al., 2013; Horn & St. Clair, 2017). In the face of a changing climate and spread of invasive species, two main drivers of wildfire severity, understanding the impacts of wildfire on animal community resilience are imperative for large-scale ecological management.

The southwestern deserts of North America are home to many endemic, at-risk species. Changes in desert fire activity may cause shifts in species composition and lead to lasting impacts on animal communities (Brooks et al., 2018). Besides the immediate impact of death and injury to wildlife, wildfires can have indirect consequences for animals, especially those of greater concern (i.e., those listed as threatened or endangered; Esque et al., 2003; Shaffer et al., 2018). Many native plants are ill-adapted to fire—if they are not completely consumed during burning, often they will die soon after—creating an opportunity for non-native grasses to

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establish following a fire disturbance (Abella, 2009; DeFalco et al., 2010; Esque et al., 2013).

Desert animals often rely on shrub habitat for foraging (Lortie et al., 2016), nesting (Kozma &

Mathews, 1997; Pidgeon et al., 2003), thermoregulation (Kerr & Bull, 2004; Ivey et al., 2020;

Lortie et al., 2022), and protection from predators (Kotler, 1984). Loss of native vegetation due

to fire and encroachment of invasive annual grasses can thus remove critical microclimates and

convert desert shrublands to grasslands with ecosystem-level effects. Loss of native vegetation

and low capacity of desert-adapted plants to recover from fire suggests that southwestern

deserts in California are sensitive to significant negative impacts from fire (Abella, 2009; Brooks,

2012; DeFalco et al., 2010).

The value of citizen science in ecological research

The need to understand wildfire impacts on desert communities and future implications

is paramount to develop effective adaptation strategies and management policies. Despite the

urgency of minimizing the impacts of large-scale environmental change on biodiversity and

ecosystem function, ecologists are faced with both limited time and resources (e.g., funding,

trained personnel) when it comes to tracking, understanding, and addressing the effects of

large-scale environmental disturbances (Williams et al., 2020). Citizen science offers a potential

source for increasing support to ecological research. Scientists can tap into citizen science

networks to meet the challenges of timely assessment of impacted biodiversity following a large-

scale environmental disturbance, like wildfires, and build capacity to assist in recovery

monitoring (Lawson et al., 2015; Kirchhoff et al., 2021). Data collected by citizen scientists (i.e.,

non-professionals) has been used to model species distributions (Renner et al., 2015), predict

species richness (Farwell et al., 2021; Carroll et al., 2022) and monitor population trends

(Neate-Clegg et al., 2020). By crowd-sourcing data collection, land managers and researchers

can better monitor ecosystem change and establish baseline data for future management

(Dickinson et al., 2012). This is not to imply that large-scale citizen science datasets replace

ecological expertise or fine-scale surveys, but that used appropriately can support monitoring and planning, especially in the wake of environmental disturbance (McKinley et al., 2017).

Recent decades have been marked by increased fire activity throughout southwestern North American deserts (Brooks & Matchett, 2006; Dennison et al., 2014; Syphard et al., 2017). Such drastic change will require land managers and conservationists to effectively and efficiently monitor biological change. Traditional ground survey work can be powerful but, at times, limited in scope—spatially and temporally. Multiple factors make survey work in deserts logistically challenging. First, more than 30% of the total land area in California is defined as a desert ecoregion (Syphard & Keeley, 2020). Second, desert fires can be large, unpredictable, and multiple burns can occur in a single season (Brooks & Matchett, 2006). Finally, key desert species have large home ranges or are highly mobile (O'Connor et al., 1994; Dickson & Beier, 2002; Cypher et al., 2013; Germano & Rathbun, 2016; Germano et al., 2021). Thus, researchers and land managers would need to employ a substantial number of personnel to cover the spatial extent of desert fires and account for varying species distribution patterns.

Citizen science and publicly accessible data through a database like the Global Biodiversity Inventory Facility (www.gbif.org) can enhance monitoring landscape-wide ecological changes and cover much larger predictive areas that would normally be difficult to survey with traditional fieldwork alone (Ivanova & Shashkov, 2021). Citizen science apps, such as eBird (www.eBird.org) and iNaturalist (www.iNaturalist.org), engage citizen scientists in wildlife observation and data collection. Citizen science is already being used to study the aftermath of fire disturbances. For example, the California Native Plant Society's Fire Followers (www.cnps.org/fire-followers) is a project coordinated through iNaturalist that encourages participants to submit observations of plants in areas burned during the 2020 and 2021 fire season. Through the efforts of more than 8,000 volunteers, over 200,000 observations have been submitted—covering a total of 46 different fires. Such observations provide valuable information in places and at scales that may not always be possible for researchers to access.

In addition to increasing spatial and temporal coverage, well-planned citizen science initiatives lend an opportunity for fine-scale observations. For instance, following the 2019-2020 unprecedented Australian bushfire season, Kirchhoff et al. (2021) were able to mobilize over 200 volunteers through iNaturalist to assess the severity and impact of fires on the over 50 million hectares of land. Citizen scientists were able to add details, such as the height at which trees burned and the percentage of vegetation scorched, to their observations that quantified burn severity at fine-scale. Such large-scale monitoring using fine-scale observations within a short time-frame (observations were collected between January 2020 and March 2020) would not be feasible without the aid of citizen scientists due to financial and personnel constraints.

Changes in wildfire regime necessitates land managers to distinguish whether shifts in community composition result from natural temporal turnover or from a more dramatic and direct response to disturbance. Long-term datasets are critical for examining temporal patterns of population dynamics and assessing the impact of disturbances on biodiversity (Magurran et al., 2010) and citizen science projects, on average, run 10 years or longer (Theobald et al., 2015). Long-running citizen science projects can collect data pre- and post- disturbance, providing important information on community response to change (Newman et al., 2017). Thus, incorporating citizen science projects in research can provide a more accurate assessment of changing ecosystems by improving both temporal and spatial coverage.

Research Objectives

This study used open-source biodiversity data to examine the impacts of fire on endangered species (ES) in California desert systems. My first objective was to examine the trend in ES occurrences reported across three major desert ecoregions from 1995 to 2020 and evaluate how these trends varied in response to fire activity. Second, I sought to evaluate the impact of fire activity on ES occurrences by comparing pre-fire and post-fire ES reporting. Finally, I estimated the post-fire recovery trend for ES across the different desert regions by

calculating a similarity index comparing ES species diversity in burned areas and areas that have no historical reports of burns as far back as the early 1900s (referred to as 'unburned', 'never-burned', or 'control' for the rest of this paper). The following questions were used to guide this study:

> (1) Has the trend in ES occurrences changed in recent decades across southwestern desert regions and are these trends influenced by fire activity?
>
> (2) Has the occurrence of fire impacted the number of ES reported?
>
> (3) Is there a long-term (21-year period) impact on ES communities in deserts post-fire and do these trends vary by desert region?

To answer these questions, aggregated ES occurrences reported on GBIF within California's southern desert ecoregions (Mojave Desert, San Joaquin Desert, and Sonoran Desert; (Germano et al., 2011; Omernik & Griffith, 2014) were contrasted between areas that have experienced fires within the 21st century (i.e. burned areas) to never-burned areas. After assessing trends in ES occurrences reported over a 26-year period across all three deserts in burned and never-burned areas, I evaluated fire-related impacts by comparing ES occurrences reported pre-fire and post-fire. Finally, I applied a linear regression to model post-fire ES recovery trends across all three deserts between 2000 and 2020.

Methods

Study area

The Mojave (MOJ), Sonoran (SON), and San Joaquin (SJD) Deserts constitute 32% (131,815 km^2) of the total land area in California (Figure 1A, Table 1). The MOJ alone accounts for more than half of this area. Desert bioregions are characterized by their low precipitation, arid soils, sparse vegetation, and desert-adapted plants and vertebrates (Whittaker, 1975;

Bailey, 2009; Brooks et al., 2018). The mean annual precipitation varies significantly within the three desert bioregions (Table 1). On the valley floors, the MOJ and SON receive an annual average precipitation ranging from 100 to 200 mm and 70 to 100 mm, respectively, but at elevations above 2,000 m the annual average precipitation ranges from 200 to 300 mm (Brooks et al., 2018). The SJD follows two decreasing moisture gradients–one from north to south, the second from east to west–and the average annual precipitation ranges from 117 mm in the southwest to 269 mm in the northeast (Germano et al., 2011; Wills, 2018). At the margins of the desert bioregions–in the leeward catchments of the mountain ranges that surround the arid lands–average annual precipitation can be as high as 600 mm (Brooks et al., 2018). Desert shrubland comprises much of the SON and MOJ–dominated by creosote bush scrub and saltbush scrub (Brooks et al., 2018). It is believed that, prior to European settlement, the SJD was also once dominated by open saltbush scrub communities but has since been converted to either dense grassland or shrubland with a dense grass understory–both dominated by non-native species (Germano et al., 2012; Wills, 2018).

The boundary layers for the MOJ and SON ecoregions were retrieved from the United States Environmental Protection Agency data repository (www.epa.gov/eco-research/ecoregions; accessed August 4, 2021). The SJD boundary layer was provided by The Nature Conservancy (Lortie et al., 2018). The SJD encompasses the western and southern two-thirds of the San Joaquin Valley as well as the adjacent Carrizo Plain and Cuyama Valley because of their unique desert flora and resident fauna (Germano et al., 2011).

Fire data were obtained from the National Interagency Fire Center (https://data-nifc.opendata.arcgis.com/; accessed August 4, 2021). Only fires that burned between 2000 and 2020 within the three desert regions and contained reports of endangered animal species occurrence were used in the study. Though some fires spread beyond the desert ecoregion, fire boundaries were clipped to only include the area within desert ecoregions. A total of 62 fires (20 in the Mojave Desert, 24 in the San Joaquin Desert, and 18 in the Sonoran Desert) ranging in

size from less than 1 km^2 to 258 km^2, with a median size of 4.3 km^2 and a mean size of 21.6 km^2 were examined (Appendix A: Table 11).

Endangered species

California Fish and Game commission lists 178 animal species under the California Endangered Species Act (California Natural Diversity Database, 2023). The list of state and/or federally endangered and threatened species was used to retrieve and compile georeferenced occurrence data from Global Biodiversity Information Facility (GBIF) on 10 May 2022 (GBIF.org, 2022) for southern California's desert ecoregions. Only 36 unique ES were reported within the desert ecoregion and kept for further analysis. Occurrences reported for the same species on the same day at identical coordinates were found to be duplicates and were removed. Only human observation or machine observation (e.g., photograph, a video, an audio recording, a remote sensing image or an occurrence record based on telemetry) records with no geospatial issues–such as null or invalid/out-of-range coordinates–were used.

A total of 24,936 animal records (Figure 1B, Table 2) for the 36 unique ES were retrieved from GBIF for the years 1995 to 2020. The occurrence records were grouped into those observed within an area that had a reported fire since the year 2000 (burned group) and those that were recorded in never burned areas (control group). The burned group included 347 reported occurrences; 160 in the MOJ, 120 in the SJD, and 67 in the SON. The control group had 24,590 reported occurrences; 8,692 in the MOJ, 9024 in the SJD, and 6874 in the SON.

Nearly all (99.6%) of the occurrence records were sourced from citizen science datasets (Table 3). The majority of the data (91%) were published by eBird, comprising almost half of all GBIF occurrences. The second largest dataset on GBIF comprised only 4% of total GBIF occurrences (Artportalen) and was not included in this study. Data collected using a standard survey protocol make up less than 0.4% (90 occurrence records) of the total dataset. Birds account for 93% of the reported individual occurrences data; therefore, analyses were

performed inclusive of birds, exclusive of birds, and only including birds to account for model sensitivity to unequal sample size between taxa groups.

NDVI data

An aggregate mean normalized difference vegetation index (NDVI) was used as a proxy for vegetation structure to examine the relationship between vegetation quality, fire activity, and occurrences reported over a 21-year period (2000-2020). NDVI values range from ≤0 (no photosynthetic activity) to +1 (high photosynthetic activity) and are derived from the ratio

$$\text{NDVI} = \frac{\text{NIR} - \text{R}}{\text{NIR} + \text{R}} \tag{1}$$

where NIR and R indicate near-infrared (841-876 nm) and red (620-670 nm) band reflectance (Myneni et al., 1995). Many studies have used NDVI as a proxy for vegetation characteristics including quantifying habitat structure (Ribeiro et al., 2019), comparing vegetative activity (Weiss et al., 2004; Horn & St. Clair, 2017; Kumari et al., 2020), estimating biomass (Casady et al., 2013), forecasting fire-risk (Michael, 2021), and assessing fire severity and post-fire recovery (Escuin et al., 2008; Esque et al., 2013; João et al., 2018). NDVI is useful for landscape ecology studies because it can be used to evaluate ecological responses to many large-scale environmental changes–including fire and other disturbances (Pettorelli et al., 2005; St-Louis et al., 2009).

Monthly NDVI data derived from MOD13C2 products (https://modis.gsfc.nasa.gov/data/dataprod/mod13.php; accessed April 19, 2022) were downloaded using the 'MODIStsp' R package (v2.0.8; Busetto & Ranghetti, 2016) and processed using the 'raster' (v3.4-10; Hijmans, 2021) and 'sf' (v1.0-7; Pebesma, 2018) R packages. Monthly NDVI raster images were used to calculate annual mean values for the years 2000-2020 (MODIS NDVI data is not available prior to 2000) at 1 km resolution. NDVI values were averaged for burned sites and for unburned sites in each desert ecoregion.

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Road Density

Primary and secondary road network data for the State of California were obtained from the United States Census Bureau (https://catalog.data.gov/organization/census-gov; accessed August 31, 2022) to explore the influence of observer bias on ES occurrence reports (Warton et al., 2013). The Line Density spatial analyst tool in ArcGIS Pro was used to calculate the density of road networks within treatment groups for each desert region (v2.9.3; Esri Inc., 2022). A linear regression was fit to test the correlation between road density and the mean annual ES occurrences reported between 1995 and 2020.

Data analysis

This study used a before/after control/impact (BACI) framework to evaluate the impact of fire disturbance on reports of ES occurrences. Reference-impact analyses between the control sites (i.e., reference condition) and the burned sites (i.e., impacted condition) enabled a contrast between reported ES occurrence in areas impacted by fires to those in areas that have historically been unaffected by wildfire to quantify the impact of fire disturbances on (van Mantgem et al., 2001). This study uses the area with no record of fire (i.e., never-burned or unburned area) of each desert region to represent control sites and desert areas that have experienced fires since 2000 (i.e., burned area) to represent the burned sites. All areas within the desert regions that have experienced a fire prior to 2000 were excluded.

To compare vegetative productivity between burned and control sites, mean annual NDVI between desert regions and treatment groups were compared using a two-way analysis of variance (ANOVA). NDVI across all three deserts was not normally distributed (Shapiro-Wilk test; p-value < 0.001). A histogram of the NDVI values shows that there are three normal peaks in the data distribution (1) one shared between the SJD burned and control group (Shapiro-Wilk test; p-value = 0.024), (2) one shared between the burned MOJ and SON (Shapiro-Wilk test; p-value = 0.494), and (3) one shared between the control group of the MOJ and SON (Shapiro-

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Wilk test; p-value = 0.077). For the first grouping, a one-way ANOVA was used to compare the effect of the treatment group on annual mean NDVI in the SJD. For the second and third grouping, one-way ANOVAs were used to compare the annual mean NDVI between the MOJ and SON ecoregions. Post-hoc Tukey HSD tests were used to evaluate which group means differed. Collectively, these tests were done to ensure that there was significant variation in NDVI between desert regions to include in the following models as a factor.

The yearly average reports of ES occurrence for the years 1995-2020 were fit to generalized linear mixed models (GLMMs). GLMMs are flexible statistical models that can handle non-normal data and allow for the incorporation of random effects (Bolker et al., 2009). The year of ES occurrence, mean annual NDVI, and treatment groups were included as an explanatory variables; however, NDVI was not a significant factor in explaining variation and, therefore, excluded from the final model. Desert ecoregion was incorporated as a random factor to remove between region variability in ES occurrence reporting (Barboza & Defeo, 2015; Foster et al., 2015). Contrasted terms were then compared using estimated marginal means. I repeated these analyses with only birds and with birds excluded to account for model sensitivity to unequal sample sizes by taxa group. The models were fit using a negative binomial distribution to account for overdispersion in the data (Stoklosa et al., 2022). ES occurrences were scaled to the number reported per 1,000 km^2 to account for the difference in size of the desert ecoregions and the differences in size between the total burned area and the unburned control area within the deserts. A generalized linear model (GLM) was fit to evaluate the effect of treatment group on the number of ES occurrences reported. GLMs are flexible models that can be fit to nonnormal data and can accommodate count data (Seavy et al., 2005).

Before/after comparisons were used to evaluate fire-related impact and analyzed using GLMMs (McDonald et al., 2000; van Mantgem et al., 2001). In this study, desert ecoregion was modeled as a random factor. ES individuals reported between 1995 and 2020 in all three deserts were used to evaluate the difference in the average of ES occurrences reported 5-years

before and 5-years after a fire. Desert region was included as a random factor. Only fires that occurred between 2000 and 2015 were considered in the burned treatment group to allow for observations 5 years before/after fire. Year of fire was used to calculate the 5-year average ES occurrences for both the burned and control group (e.g., to evaluate the impact of the 2005 Paradise fire in the MOJ, the average the ES species occurrences reported between 2000-2004 and the ES occurrences reported between 2006-2010 were used to calculate the 5-year average ES occurrences before and after a fire, respectively. Similarly, the ES species occurrences reported between 2000-2004 and the ES species occurrences reported between 2006-2010 in the MOJ historically unaffected by fire were used to calculate the 5-year average ES occurrences before and after, respectively, for the control group). These analyses were repeated for excluding avian species occurrences as well as only avian species occurrences to account for the skew of this overrepresented animal class. ES occurrences are scaled to the number of individuals reported per 10,000 km^2 to account for the difference in size of the desert ecoregions and the difference in size between the total burned area and the unburned control area within the deserts. Models were checked for overdispersion and zero-inflation (Ver Hoef & Boveng, 2007; Zuur et al., 2009).

An incidence-based ChaoSørensen index comparing burned:control ES composition was calculated for all three deserts to evaluate the change in post-fire ES composition through time. The equation for incidence-based ChaoSørensen index is

$$\text{ChaoSørensen}_{incidence} = \frac{2UV}{U+V} \tag{2}$$

where U denotes the total number of shared species between treatment groups that were reported in the burned group and V represents the total number of shared species that were reported in the unburned group. Both U and V account for unseen (i.e. unreported) shared species using observed rare species to estimate an adjustment term (Chao et al., 2005). This

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metric tests for differences in community composition because it is an adjusted similarity index that reduces sample-size bias. Only three years, four years, and seven years in the MOJ, SON, and SJD, respectively, during the 21-year period had sufficient data for estimating the ChaoSørensen index. To analyze the trajectory in burned composition, a linear regression was applied to test for the effect of time since fire on the similarity index for community composition (Abella, 2009).

Data retrieval, data processing, statistical analysis, and visualizations were done using R (v4.2.0; R Core Team, 2022). The 'glmmTMB' package (v1.15; (M. E. Brooks et al., 2017) was used to fit GLMMs. All models were tested for overdispersion (using the check_overdispersion function) and zero-inflation (using check_zeroinflation function) in the 'performance' package (v0.9.0; Lüdecke et al., 2021) and were fit to either a quasi-Poisson or negative binomial distribution as appropriate (Ver Hoef & Boveng, 2007). The 'emmeans' package (v1.8.4-1; Lenth et al., 2023) was used to compute contrasts of estimated marginal means. The 'SpadeR' package was used to calculate the incidence-based ChaoSørensen index (v0.1.1; Chao et al., 2016). Spatial processing of occurrence records and fire sites was completed using ArcGIS Pro (v2.9.3; Esri Inc., 2022).

Results

Differences in mean annual NDVI

Separate ANOVA results were estimated for SJD, MOJ and SON (inclusive of both treatments), burned treatment only of MOJ and SON, and control treatment only of MOJ and SON due to the distribution of the data. The mean annual NDVI (Figure 2; Table 4) was significantly different between treatment groups (ANOVA[1], $p < 0.001$) and desert ecoregions (ANOVA[1], $p < 0.001$). Between desert comparison showed that mean annual NDVI was greater

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[1] Does not meet normality assumption

in SJD than both the MOJ (Tukey HSD, difference = 0.175, $p < 0.001$) and the SON (Tukey HSD, difference = 0.165, $p < 0.001$). There was no significant difference of mean annual NDVI between MOJ and SON burned treatments (Tukey HSD, difference = 0.002, $p = 0.819$); however, the SON had a slightly higher mean annual NDVI in the control treatment than the MOJ (Tukey HSD, difference = 0.017, $p < 0.001$).

Between treatment group comparison showed that the mean annual NDVI was higher in the unburned area of SJD than the burned area (Tukey HSD, difference of 0.02, $p = 0.018$). Unlike the SJD, both the MOJ and SON had higher mean annual NDVI at burned sites than at control sites (ANOVA[2], $p < 0.001$; Tukey HSD, difference of 0.097, $p < 0.001$).

Differences in ES occurrences reporting

Over the 26-year period (1995-2020), both treatment groups showed an overall positive trend in the yearly average ES occurrences reported for all three deserts (Figure 3;

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[2] Does not meet normality assumption

Table 5). Although, sensitive analyses for the effect of birds were consistent with the main model, there was a greater positive trend in the number of annual non-avian ES occurrences reported in the burned treatment than in the control treatment (post-hoc contrast, p < 0.001), a pattern not seen when only modeling the reports of avian occurrences. The more positive trend in non-avian occurrence reported in the burned treatment is likely skewed by the relatively high number of desert bighorn sheep (*Ovis canadensis*) and mountain lions (*Puma concolor*) reported in the SON. There were 23 reports of desert bighorn sheep and 13 reports of mountain lions in the SON; the highest number of any non-avian species reported in other deserts was 8 (MOJ; desert gopher tortoise, *Gopherus agassizii*) and 3 (SJD; San Joaquin antelope squirrel, *Ammospermophilus nelsoni*).

In the 26-year period, more ES occurrences per 1,000 km^2 were reported in burned treatment group than in the control treatment group (Figure 2a;

Table 6; post-hoc contrast, p < 0.001).The reported avian ES occurrences in the SJD were significantly greater than all other desert regions (Table 7; post-hoc contrast, p < 0.001); however, this was not the case for non-avian reports—which had the greatest number reported in the SON (post-hoc contrast, p < 0.001).

There was a positive correlation between road density and the number of ES occurrences reported (Figure 6b, p = 0.073). There was no significant difference in the road density between treatment groups; however, the burned treatment in the SON had the greatest road density of all deserts and treatment groups (median = 27.47 m of road/km^2; Figure 6a) and the MOJ had the lowest median road density of all three deserts, regardless of treatment group (burned = 7.42 m of road/km^2; control = 6.92 m of road/km^2).

Impact of fires on ES occurrences

There were significantly more ES occurrences reported in the 5-year period following a fire than the 5-year period preceding a fire, regardless of treatment group (Figure 4; Table 8;

GLMM, $p = 0.003$; post hoc contrast, $p = 0.003$). In other words, reports of ES occurrences increased in future years regardless of fire activity. Additionally, there was a significantly greater number of ES occurrences reported in the control group than in the burned group, both pre- and post-fire (post hoc contrast, $p < 0.001$). This contradicts the results in previous model, highlighting that when accounting for the specific year a fire occurred the control group had more reports of ES occurrences than the burned group. Sensitivity analyses for the effect of birds were consistent with the main model (GLMM, $p = 0.001$, post-hoc contrast, $p = 0.001$).

Recovery of ES post-fire using community composition measures

There were no significant trends for ES composition in burned areas resembling unburned areas with increasing time since fire (Figure 5;

Table 9; LM, $p = 0.512$. Among the three deserts, the SON had the lowest similarity between burned:unburned ES composition (30±10% [ChaoSørensen similarity mean ± standard error], post-hoc contrast, $p = 001$). The MOJ had the greatest similarity between burned:unburned ES composition (95±10%), there was no significant difference between the MOJ and SJD burned:unburned ES composition (post-hoc contrast, $p = 0.484$). Insufficient data made it difficult to estimate the ChaoSørensen similarity index for the entirety of the study period; however, ChaoSørensen similarity estimates were available for 4-19 years since fire– SJD having the longest time coverage. Thus the similarity in ES composition between burned and unburned sites varied across desert regions.

Discussion

Key findings

The aim of this study was to examine the impacts of fire on endangered species in California desert systems. The results show that fire activity in the desert had a significant effect on the 36 ES that were reported within these regions. Model sensitivity analyses show that

avian and non-avian species were similarly impacted by fire disturbance. Findings suggest that burned habitats are still able to support ES communities and that ES continue to visit or occupy burned desert areas. However, never-burned sites had greater reports of ES occurrences in the time following fire and the increase in reports of ES occurrences post-fire in burned habitat is likely influenced by increased participation of citizen scientists—due to increased adoption of reporting apps by the general public. Although BACI analyses show less ES reported in burned areas than the control, high similarity in ES composition between the treatment groups in the MOJ and SJD suggest that diversity of ES is not always affected by fire. Additionally, in the 19 years following fire, the similarity in ES composition between burned and unburned desert sites remained unchanged. Differences in mean annual NDVI between burned and control groups supports that California's arid landscapes are composed of varying vegetative heterogeneity and that fires were more likely to occur in areas with greater plant biomass. The findings here support the use of GBIF data as a tool to explore and monitor landscape-level analysis of ES responsiveness to fire, but that doing so should include measures to reduce sampling bias that may lead to false narratives.

Endangered animal species show resilience to fire disturbance

This study shows that endangered animal communities continue to occupy burned landscapes both short-term and long-term. The impact of fires on ES in southwestern deserts was examined by comparing the number of occurrences reported before and after a fire in burned and unburned areas throughout the MOJ, SON, and SJD. There was a higher number of ES occurrences reported post-fire than pre-fire in both treatment groups, which suggests that fires do not appear to have an net negative impact on ES occurrences across the three desert systems. In fact, some deserts showed a higher relative number of ES occurrences reported in post-fire burned areas compared to the control. Fires in desert lands can play a significant role in transforming desert landscapes and fire-mediated habitat change can influence population

dynamics of ES through various mechanisms (Abella et al., 2009; Clapp & Beck, 2016; Dorph et al., 2020; Geary et al., 2020).

Studies suggest that fire disturbances can improve habitat for desert species that favor open spaces (Monasmith et al., 2010; Horn et al., 2012). Abella et al. (2009) shows that fires can reduce the coverage of some invasive grass species (e.g., *Bromus rubens*) and promote the recovery of native shrublands–as long as native plants are not completely consumed by the fire. Many desert ES rely on the open ground of sparsely vegetated shrublands to forage and avoid predation (Germano et al., 2001; Inman et al., 2013), including the following species: blunt-nosed leopard lizards (*Gambelia sila*), giant kangaroo rats (*Dipodomys ingens*), Tipton kangaroo rats (*Dipodomys nitratoides nitratoides*), San Joaquin antelope squirrels (*Ammospermophilus nelsoni*), and Mohave ground squirrels (*Xerospermophilus mohavensis*). Some research also suggests that upstream and upland fires can improve aquatic habitats for aquatic and amphibian ES by creating new essential habitat elements from woody debris (e.g., pools, complex structures, stabilized banks; Rieman et al., 2012). Breeding habitat for arroyo toads (*Anaxyrus californicus*) are thought to improve when stream beds are cleared of dense, decadent vegetation and restored with coarse sediment deposits following a fire (Mendelsohn et al., 2005). In this study, only burned areas contained reports of arroyo toads, supporting the prediction that fires can improve some riparian habitats for ES. Conway et al. (2010) have shown Yuma Ridgway's rails (*Rallus obsoletus yumanensis*) can also respond positively to cleared dense vegetation in burned wetland habitat; however, in this study there were no reports of Yuma Ridgway's rails in burned sites after fire, despite there being a greater number of occurrences reported at burned sites prior to fire than at never burned sites. Moreover, fire-mediated habitat change can indirectly influence population dynamics through predator-prey interactions. Desert bighorn sheep (*Ovis canadensis nelsoni*) not only benefit from increased forage quality following a fire, but also increased visibility that allows them to more easily detect predators (Holl & Bleich, 2012). Several studies have shown that some predators may be

attracted to recently burned areas because of increased prey availability (Geary et al., 2020; Doherty et al., 2022). Simultaneously, predators may benefit from reduced habitat cover—which leaves some prey species more vulnerable—and decreased competition from larger, ambush predators (Warrick & Cypher, 1998; Geary et al., 2020; Doherty et al., 2022). This is in agreement with my findings that show a relatively high number of desert bighorn sheep and mountain lion (*Puma concolor*) occurrences reported in the burned areas of the SON in this study. Thus, fire-mediated habitat change can directly and indirectly impact at-risk species that occupy deserts; therefore, it is imperative for land managers to be able to effectively monitor the impacts of fire disturbance on desert systems.

Not all fire-impacted desert lands remain as the open sparsely-vegetated shrublands preferred by many desert-adapted endangered species. Although this study did not compare plant communities pre- and post-fire, other studies have shown that some non-native grasses respond positively to fire and can out-compete native forbs during the recovery period post-disturbance–transforming the vegetative composition of desert ecosystems (St. Clair et al., 2016; Bishop et al., 2020). Consequently, Abella (2009) shows that even more than forty years following a fire, plant composition in burned areas continues to be significantly different than plant composition in unburned areas. Most prior research supports a decrease in native plant diversity in response to fire disturbance (Steers & Allen, 2011, 2012; Horn & St. Clair, 2017; Underwood et al., 2019). It is clear that the influence of fire on native vegetation is complex and, as a result, the impacts of fire disturbance on ES is not as straightforward. For example, while quadrupedal rodents prefer shrub cover–which may be consumed by fire, bipedal rodents, such as kangaroo rats (*Dipodomys* spp.), benefit from the increased open space created by fires (Horn et al., 2012; Sharp Bowman et al., 2017; Bishop et al., 2020; Chock et al., 2020). Moreover, post-fire rodent activity can influence the establishment of invasive grasses through seed predation and herbivory (Bishop et al., 2020). While this study highlights a general

resilience of ES to desert fires, understanding the source of that resiliency for specific species can aid in conservation planning.

There are multiple mechanisms to explain the pattern in ES persistence following fire disturbance. Despite a loss in native plant diversity, burned areas can recover shrub coverage and density in the years following fire (Steers & Allen, 2011; Horn et al., 2015), providing essential habitat for animals that occupy the area. The ability of post-fire habitats to continue to support native endangered animal communities in the long-term might be an outcome of fire nature in desert ecosystems. Most desert fires are patchy and of low intensity, allowing plants to survive in unburned islands and animals to seek refuge in unburned microhabitats that can serve as shelter from predation post-fire (McLaughlin & Bowers, 1982; Pausas, 2019). In the long-term, unburned refugia can serve a critical role for landscapes recovering from fire disturbance–serving as a source from which plants and animals expand to re-colonize recovering habitat (Meddens et al., 2018; Mendelsohn et al., 2008). Animals are able to modify their habitat-use rather than simply avoid or seek burned landscapes (Pastro et al., 2011; Saracco et al., 2018; Dorph et al., 2020; Geary et al., 2020). Desert ectotherms, such as blunt-nosed leopard lizards and desert tortoises (*Gopherus agassizii*)–endangered species endemic to southern California's arid region, rely on shade by shrub canopy for thermoregulation (Drake et al., 2015; Ivey et al., 2020). Both have been documented to continue using dead structures for cover and shade following a fire (Drake et al., 2015; Germano, 2019). In the absence of shrubs, blunt-nosed leopard lizards have shown to rely on burrows to escape the heat during the day (Ivey et al., 2020; Zuliani et al., 2023) –which may be important for their continued presence in sites that have experienced repeated fire activity and, therefore, lack live or dead shrub cover. Avian species that rely on native shrubs and plants as important sources of seed-food, cover from predators, and nesting sites (Bock & Block, 2005) are highly mobile and can move between burned and unburned patches to meet their resource demands (Mendelsohn et al., 2008; Martin & Fahrig, 2018). As a result, ES response to fire might display a temporal lag

as species shift away from burned areas and then shift-back during recovery. For this reason, a 5-year period before and after fire was used to try to capture the lag in animals return to burned landscapes. My results imply that the use of habitat by ES post-fire is viable and that arid lands affected by fire can still support ES, which is promising information for managers seeking to protect at-risk species in burned areas.

Resiliency varies between desert ecoregions

Notably, precipitation levels may explain some of the differences seen in ES composition differences between deserts. The SON is the most arid of the three desert regions explored in this study and is the least impacted by invasive grasses (Brooks & Pyke, 2001; Moloney et al., 2019). Thus, the relatively low similarity in ES composition between burned and unburned sites in the SON might be explained in two parts: (1) invasive grasses historically first arrived to–and more easily established in–the more mesic regions of the SJD and MOJ, giving those ecosystems more time to acclimate (Brooks & Pyke, 2001; Germano et al., 2001; Moloney et al., 2019); and (2) increased fire activity in the SON and the bimodal rain season might be promoting recent increasing invasive grass establishment, especially in years of higher rainfall (Brooks & Pyke, 2001; Moloney et al., 2019). Thus, given more time, ES composition in SON burned regions might begin to resemble more closely to unburned regions.

The SJD is an example of a desert to have once been dominated by saltbush scrub communities mixed with patches of native grasses, but the invasion of non-native grasses has created an unrecognizable landscape dominated by either dense grassland or shrubland with a dense non-native grass understory (Kelly et al., 2006; Germano et al., 2011, 2012; Wills, 2018). This study showed that the SJD had the highest mean annual NDVI among the three deserts, moreover, the mean annual NDVI did not differ between burn and control treatment groups. Unlike the MOJ and SON, the SJD has experienced significant human disturbance (e.g., intensive agriculture, oil extraction, and urban development) since European settlement of the

region (Kelly et al., 2006; Germano et al., 2011). Higher annual NDVI values and greater NDVI

variability, like those seen in the SJD, has been linked to increased non-native grass density

(Horn & St. Clair, 2017), which in turn creates a continuous fuel bed not typical in barren desert

landscapes. Burned sites in the MOJ and SON exhibited greater vegetative biomass than

control sites. Most fires in the MOJ and SON occurred on the western edge of the desert

ecoregions and at higher elevations (Brooks et al., 2018), where precipitation–and plant

productivity–is greatest due to the orographic lift created by the mountain ranges that bound the

desert regions (Minnich, 2018). This supports findings in previous studies that show increased

fire risk in deserts are associated with greater biomass accumulation (Brooks & Matchett, 2006;

Balch et al., 2013; Casady et al., 2013; Klinger et al., 2021) and probability of large fires

increase with greater NDVI (Gray et al., 2014). Moreover, fires can further perpetuate the

invasion of desert shrublands by non-native grasses (Germano et al., 2001). Fires can reduce

native shrub cover and density by as much as 90% in post-fire desert landscapes compared to

unburned landscapes (McLaughlin & Bowers, 1982; Horn & St. Clair, 2017), making burned

landscapes more susceptible to non-native grass invasion (Villarreal et al., 2016; Underwood et

al., 2019). The conversion of desert shrubland to grassland reduces optimal habitat available for

many desert-adapted at-risk species (Germano et al., 2001). Although this study shows that ES

are still present in burned environments, continued monitoring is needed to gain a deeper

understanding on how projected changes in fire regimes might impact at-risk species in the

future.

The cost of citizen science

Citizen science can play a key role in assessing and monitoring ecosystem-wide change

that occurs following a fire disturbance. It has been estimated that over a million citizen

scientists have contributed over half a billion dollars annually of in-kind donation through

volunteer labor on biodiversity projects (Theobald et al., 2015). Ecologists can expect to see

increased implementation of citizen science within research projects. The last several decades have already seen a significant growth in the number of biodiversity-focused citizen science projects (Theobald et al., 2015). Moreover, changes in policies and grant requirements further incentivize the use of citizen science in research projects. In 2017, the United States passed the Crowdsourcing and Citizen Science Act (15 U.S.C. § 3724), which encourages agencies to use citizen science in scientific research. The National Science Foundation (2023) –a major research funder in the United States–now requires public engagement as a condition to receive funding. Both eBird and iNaturalist have experienced a steady exponential increase in users and observation submission since their launch, in 2002 and 2008, respectively (Loarie, 2020; Team eBird, 2021). The increasing trend of ES occurrences reported in all three desert regions and across both burned and unburned sites within the last 21 years reflects the growing use of citizen science data collection. GBIF is an omnium gatherum of biodiversity datasets, and citizen science data submissions–specifically observations submitted through eBird and iNaturalist–make up a majority of species occurrence records stored on GBIF. The inevitable expansion and universal implementation of citizen science in ecological research, requires critical challenges to be addressed in order to effectively incorporate citizen science data into projects.

This study, like all studies that rely on citizen science data, is prone to sampling bias. Citizen science is valued as a tool capable of filling spatial gaps in ecological research; however, data collection by citizen scientists tends to be opportunistic–collected without standardized field protocols or explicit sampling design– which, by nature, is unstructured (van Strien et al., 2013; Bayraktarov et al., 2019; Hughes et al., 2021). Non-professionals show a preference for collecting observations in easier to reach and more desirable locations–largely concentrated around major urban areas and in close proximity to roads, coasts, and shipping routes–perpetuating gaps in spatial coverage (Warton et al., 2013; Fithian et al., 2015; Lloyd et al., 2020; Hughes et al., 2021; Ivanova & Shashkov, 2021). Remote areas, like those common

in deserts, tend to be under-sampled (Hughes et al., 2021). In this study, burned areas generally contained greater road density and there was a positive correlation between road density and mean occurrences reported; therefore, the greater prevalence of ES occurrences in burned areas could be impacted by the greater accessibility to burned sites compared to unburned desert areas. On the other hand, citizen scientists might display avoidance of areas that are perceived as less desirable for species observation (i.e., observers think they are unlikely to encounter interesting species and, therefore, choose to go to other sites), like those recently impacted by fires (Johnston et al., 2022). The number of times sites are visited for sampling may have minimal impact on trend estimates (Isaac et al., 2014); however, the burned sites available in this study for use in fire impact modeling were limited by the number of burned sites visited both before and after fire. Additionally, most citizen science apps explicitly record positive observations (i.e., presence-only), which limits the application of more powerful statistical analyses (Isaac et al., 2014). Complete checklists (i.e., lists that contain every species observers were able to identify to their best effort) can be used to generate presence-absence data. Although some apps, like eBird, already allow for observers to submit complete checklists, wider implementation of complete checklists in citizen science can make it a more powerful research tool. Furthermore, citizen science reporting shows a bias towards rare (i.e., at-risk) species reporting (i.e., rare species are more interesting to citizen scientists), which can produce a false narrative that endangered species populations are improving when, in reality, they are simply overreported (Boakes et al., 2010; Lloyd et al., 2020; Johnston et al., 2022). Lloyd et al. (2020) report that the density of citizen science observations increase in areas with greater at-risk species richness. The ChaoSørensen similarity index, used in this study, is a statistical tool that reduces bias in biodiversity measures by weighing the frequency and identity of uncommon species to account for the effect of unseen (i.e., unreported) shared species between two communities (Chao et al., 2005). Additionally, combining citizen science datasets with professional datasets or ground-truthing results can improve the capacity of citizen science

data to monitor environmental change (Crall et al., 2015). Thus successful implementation of citizen science data relies on the development of robust statistical methods that can account for observer bias to reduce misinterpretation of ES population dynamics (Bird et al., 2014; Isaac et al., 2014; Bayraktarov et al., 2019; Johnston et al., 2022).

Land managers and researchers can improve habitat disturbance monitoring by incorporating the use of citizen science data (Crall et al., 2015; Kirchhoff et al., 2021). The vastness of California's southwestern desert systems as well as the unpredictability of wildfires presents a challenge for conservation managers. Managers can incorporate the use of citizen scientists to quickly and effectively collect information on fire severity and impact (Kirchhoff et al., 2021). In addition to monitoring post-disturbance changes, land managers can proactively engage citizen scientists to collect information on species presence to establish a baseline measure of biodiversity. This will help with ES monitoring efforts because it would allow for before-/after- disturbance comparisons. Several frameworks have already been suggested with regards to incorporating citizen science to target spatial and temporal gaps. Callaghan et al. (2019) propose incentivizing volunteers to sample in particular places and times (e.g., programs such as the Christmas Bird Count; (Meehan et al., 2019) instead of focusing on particular species or number of species recorded, as well as evaluating research sites for their high-marginal-value (e.g., those at greater fire-risk) and targeting those areas. Projects should leverage the 'power-of-place'–that is, the emotional, cultural, and material connection that many people have for the place they live in–to increase participation and data utilization (Newman et al., 2017). This includes working with local communities to co-create projects that address public interests and abilities, as well as providing input, resources, and commitment from scientists on projects that are community driven and organized (Newman et al., 2017). Scientists can leverage their expertise in community-focused projects by helping train volunteers and supporting best practices to ensure quality data and interpretations made with high scientific integrity. The intentional implementation of citizen science can provide more

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complete biodiversity information relevant for land management, especially in the face of a rapidly changing environment.

Conclusions

The potential for future fire activity to increase in California desert systems will require novel tools to efficiently and effectively monitor ecological change over a large expansive area (Brooks et al., 2004; Brooks & Zouhar, 2008; Balch et al., 2013; Underwood et al., 2019). This work used open-source data—mostly collected by citizen scientists—from GBIF to understand the impacts of fires in the 21st century on endangered species in the Mojave, San Joaquin, and Sonoran desert regions. Regardless of fire activity, all desert regions experienced a general trend of increasing endangered species occurrences over the last 25 years. The higher number of ES reported post-fire was surprising since fires are associated with conversion of shrublands to grasslands and, therefore, the loss of important desert habitat (Brown & Minnich, 1986; D'Antonio & Vitousek, 1992; Germano et al., 2001; Brooks & Esque, 2002). The continued presence of at-risk species in burned habitat may provide a window of opportunity to land managers seeking to protect them. Land managers can mediate the long-term impacts of fire disturbances through invasive grass management and promoting the recovery of native vegetation (Diffendorfer et al., 2012). The use of frequent prescribed burns to clear denser vegetation, control invasive grass species, can promote suitable open habitat for at-risk species (Brooks & Pyke, 2001; Germano et al., 2001; Salvatori et al., 2001; Conway et al., 2010; Green et al., 2015; Clapp & Beck, 2016). In addition to promoting suitable habitat for at-risk species that favor open spaces, prescribed burns can reduce the availability of continuous fine fuels–reducing risk of larger and more severe future burns. This study shows that desert animals continue to occupy burned environments, and yet burned habitat take many years to recover and regain plant cover. Land managers can provide essential habitat structures to alleviate

some of the negative impacts of fire. For example, desert animals that rely on shrubs for

thermoregulation may benefit from deployed artificial shelters that can temporarily serve as

thermal refuges for animals while vegetation recovers (Drake et al., 2015; Ghazian et al., 2020;

Ivey et al., 2020). Microclimates simulated by artificial shelters can also of promote the recovery

of native vegetation (Filazzola & Lortie, 2014; Ghazian et al., 2020; Lortie et al., 2022). By

supporting endangered animal species that continue to visit post-fire landscapes, managers

may ameliorate the negative impacts of fire disturbance in desert landscapes.

Tables and Figures

Table 1: Description of desert bioregions examined in this study. Total area represents that area that is found within the state of California. Total area burned includes only fire sites which also had reports of ES. Total ES occurrences reported per 1,000km^2 in burned and never-burned area.

Desert region[a]	Total Area (km^2)	Mean annual precipitation at valley floors (mm)[b]	Total area burned (km^2) 2000-2020	Total ES occurrences reported per 1,000 km^2 1995-2020
Mojave	73,883	100-200	722	343
San Joaquin	28,008	117-269	219	643
Sonoran	26,936	70-100	399	556

NOTES:

[a](Germano et al., 2011; Omernik & Griffith, 2014)

[b](Brooks et al., 2018; Germano et al., 2011; Wills, 2006)

Table 2: List of endangered animal species retrieved in occurrence data acquired from GBIF. Asterisk (*) following species name indicates avian species.

| Species | Common Name | Federal Status | State Status | Total number of occurrences reported within desert region | | | | | | Total |
| | | | | Mojave | | San Joaquin | | Sonoran | | |
				Burned	Control	Burned	Control	Burned	Control	
*Agelaius tricolor**	Tricolored blackbird		ST	13	1275	24	2840	1	317	4470
Ambystoma californiense	California tiger salamander	FE	ST				1			1
Ammospermophilus nelsoni	Nelson's antelope squirrel		ST			3	107			110
Anaxyrus californicus	Arroyo toad	FE		2						2
Batrachoseps stebbinsi	Tehachapi slender salamander		ST		1					1
Bombus crotchii	Crotch bumble bee		SC		6		6	1	3	16
*Branta hutchinsii**	Cackling goose	FDR					4			4
*Buteo swainsoni**	Swainson's hawk		ST	13	1158	26	4094	4	1114	6409

Species	Common Name	Federal Status	State Status	Total number of occurrences reported within desert region						Total
				Mojave		San Joaquin		Sonoran		
				Burned	Control	Burned	Control	Burned	Control	
*Colaptes chrysoides**	Gilded northern flicker		SE	74	86				27	187
Coleonyx switaki	Barefoot gecko		ST						3	3
Cyprinodon macularius	Desert pupfish	FE	SE					8	13	21
Danaus plexippus	Monarch		FC	3	32	1	18		50	104
Dinacoma caseyi	Casey's June beetle	FE					1		3	4
Dipodomys ingens	Giant kangaroo rat	FE	SE				17			17
Dipodomys nitratoides	Tipton kangaroo rat	FE	SE				2			2
*Empidonax traillii**	Willow flycatcher		SE	36	4047	10	453	6	1092	5644
Euproserpinus euterpe	Kern primrose sphinx moth		FT				1			1
*Falco peregrinus**	American peregrine falcon	FDR	SDR						1	1

Species	Common Name	Federal Status	State Status	Total number of occurrences reported within desert region						Total
				Mojave		San Joaquin		Sonoran		
				Burned	Control	Burned	Control	Burned	Control	
Gambelia sila	Blunt-nosed leopard lizard	FE	SE				49			49
Gopherus agassizii	Desert tortoise	FT	ST[1]	8	412			1	127	548
*Gymnogyps californianus**	California condor	FE	SE		8	5	442			455
*Haliaeetus leucocephalus**	Bald eagle	FDR	SE	3	117	50	815		166	1151
*Melanerpes uropygialis**	Gila woodpecker		SE		6			4	2849	2859
*Micrathene whitneyi**	Elf owl		SE						3	3
Ovis canadensis	Peninsular bighorn sheep	FE	ST	2	113			23	433	571
*Pelecanus occidentalis**	California brown pelican	FDR	SDR				1		10	11
Puma concolor	Mountain lion		SC	1	2			13	32	48
*Rallus obsoletus**	Yuma Ridgway's rail	FE	ST					1	21	22
Rana boylii	Foothill yellow-legged frog		SE				3			3

Species	Common Name	Federal Status	State Status	Total number of occurrences reported within desert region						Total
				Mojave		San Joaquin		Sonoran		
				Burned	Control	Burned	Control	Burned	Control	
Rana draytonii	California red-legged frog	FT			1		1			2
Rana muscosa	Southern mountain yellow-legged frog	FE	SE		1					1
*Riparia riparia**	Bank swallow		ST	5	1403	1	149		483	2041
Uma inornata	Coachella Valley fringe-toed lizard	FT	SE		15			5	124	144
*Vireo bellii**	Least Bell's vireo	FE	SE		5				3	8
Vulpes macrotis	San Joaquin kit fox	FE	ST				21			21
Xerospermophilus mohavensis	Mohave ground squirrel	FT	ST		2					2
			Total	**160**	**8691**	**120**	**9024**	**67**	**6874**	**24936**

FE = Federally listed endangered; FT = Federally listed threatened; FDR = Federally delisted (recovered)
SE = State listed endangered; ST = State listed threatened; SC = State candidate for listing; SDR = State delisted (recovered)
[1] *Gopherus agassizii* is currently listed as threatened in California; however, since 2020 it has been a state candidate for endangered listing.

Table 3: List of datasets involved in GBIF report. All, except the Avian Knowledge Network, are datasets compiled by citizen science.

GBIF dataset	Survey protocol	Total ES occurrences
eBird	Citizen Science	22,682
iNaturalist	Citizen Science	2,072
Avian Knowledge Network – Point Reyes Bird Observatory	Standard survey – Point Counts	90
Great Backyard Bird Count	Citizen Science	39
Xeno-canto	Citizen science	28
eButterfly	Citizen science	7
The Cornell Lab of Ornithology – Macaulay Library	Citizen science	6
Observation.org	Citizen Science	5
The Lepidopterists' Society	Citizen Science	4
Naturgucker.de	Citizen Science	3

Table 4: Results of analysis of variance (ANOVA) of the effect of desert region and treatment group on the mean annual normalized difference vegetation index (NDVI) from 2000-2020 in the Mojave, Sonoran, and San Joaquin deserts. Mean annual NDVI values inclusive of all three deserts were not normally distributed; therefore, separate ANOVA results were estimated for the SJD burned and control treatment, SON and MOJ burned treatment group, and SON and MOJ control treatment group each meeting the assumption for normal distribution. All significant p-values ($p < 0.05$) are indicated in bold.

Response	Source	df	Sum Sq	Mean Sq	F-value	p-value
NDVI[†]	Desert region	2	0.812	0.406	290.84	**< 0.001**
	Treatment group	1	0.100	0.100	71.45	**< 0.001**
NDVI *SJD*	Treatment group	1	0.006	0.006	6.099	**0.018**
NDVI[†] *SON + MOJ*	Treatment group	1	0.196	.0196	551.6	**< 0.001**
NDVI *Burned: SON + MOJ*	Desert region	1	<0.0001	<0.0001	0.053	0.819
NDVI *Control: SON + MOJ*	Desert region	1	0.003	0.003	17.13	**<0.001**

[†]Data is non-normal

Table 5: Analysis of trend in yearly average reported ES occurrences between treatment groups from generalized linear mixed model (GLMM) for 1995-2020. Desert region was included as random effect. All significant p-values ($p < 0.05$) are indicated in bold.

		df	Chi-square	p-value
All ES	(Intercept)	1	72.75	**<0.001**
	Year	1	75.06	**<0.001**
	Treatment group	1	13.25	**<0.001**
Avian ES	(Intercept)	1	231.72	**<0.001**
	Year	1	237.08	**<0.001**
	Treatment group	1	1.31	0.252
Non-avian ES	(Intercept)	1	54.12	**<0.001**
	Year	1	54.82	**<0.001**
	Treatment group	1	33.86	**<0.001**

Table 6: Analysis of mean annual ES occurrences reported in response to desert region and treatment group from generalized linear model (GLM) for 1995-2020. All significant p-values ($p < 0.05$) are indicated in bold.

		df	Deviance	Residual df	Residual deviance	p-value
Avian ES	NULL			125	163.68	
	Desert region	2	25.40	123	138.28	**<0.001**
	Treatment group	1	6.64	122	131.65	**0.010**
	Desert:Treatment group	2	1.78	120	129.87	0.411
Non-avian ES	NULL			75	223.651	
	Desert region	2	103.63	73	120.02	**<0.001**
	Treatment Group	1	65.05	72	54.97	**<0.001**
	Desert:Treatment group	2	5.36	70	49.61	0.068

Table 7: Results from emmeans post hoc test for the generalized linear model analyzing the response of endangered species occurrence reporting between deserts and treatment groups. Main effects, desert region and treatment groups, were tested separately because interaction between desert regions and treatment groups was insignificant. All significant p-values ($p < 0.05$) are indicated in bold.

	Main effects	Response	SE	df	z ratio	p-value
	Desert					
Avian ES	Mojave	6.49	0.769	Inf	15.7654	**<0.001**
	San Joaquin	15.82	1.954	Inf	22.358	**<0.001**
	Sonoran	9.33	1.518	Inf	13.718	**<0.001**
Non-avian ES	Mojave	1.96	0.358	Inf	3.705	0.883
	San Joaquin	2.28	0.571	Inf	3.277	**<0.001**
	Sonoran	7.20	0.997	Inf	14.238	**<0.001**
	Treatment group					
Avian ES	Control	8.33	0.775	Inf	22.805	**<0.001**
	Burned	11.66	1.482	Inf	19.318	**<0.001**
Non-avian ES	Control	1.57	0.213	Inf	3.296	**0.001**
	Burned	6.45	1.167	Inf	10.311	**<0.001**

Table 8: Analysis of ES occurrences reported in the 5-year period pre-fire and the 5-year period post-fire from the generalized linear mixed model (GLMM) for 1995-2020. All significant p-values ($p < 0.05$) are indicated in bold.

		df	Chi-square	p-value
All ES	(Intercept)	1	0.0003	0.986
	Pre/post	1	8.529	**0.003**
	Treatment group	1	159.227	**<0.001**
	Pre/post:Treatment group	1	3.544	0.060
Avian ES	(Intercept)	1	0.903	0.342
	Pre/post	1	5.516	**0.019**
	Treatment group	1	112.157	**<0.001**
	Pre/post:Treatment group	1	1.075	0.300
Non-avian ES	(Intercept)	1	2.363	0.124
	Pre/post	1	12.185	**<0.001**
	Treatment group	1	28.102	**<0.001**
	Pre/post:Treatment group	1	1.751	0.185

Table 9: Analysis of ChaoSørensen similarity index for ES composition in burned and unburned sites from linear model (LM). Similarity index estimated for 4-19 years since fire. All significant p-values ($p < 0.05$) are indicated in bold.

		df	Sum Sq	Mean Sq	F value	p-value
Avian ES	Time since fire	1	0.017	0.017	0.457	0.512
	Desert	2	0.985	0.492	13.149	**0.001**

Table 10: Post hoc test results for LM of ChaoSørensen similarity index for ES composition in burned and unburned sites. All significant p-values ($p < 0.05$) are indicated in bold.

Desert	Emmean	SE	df	t ratio	p-value
Mojave	0.947	0.100	12	9.475	**<0.001**
San Joaquin	0.739	0.082	12	9.032	**<0.001**
Sonoran	0.297	0.092	12	3.230	**0.007**
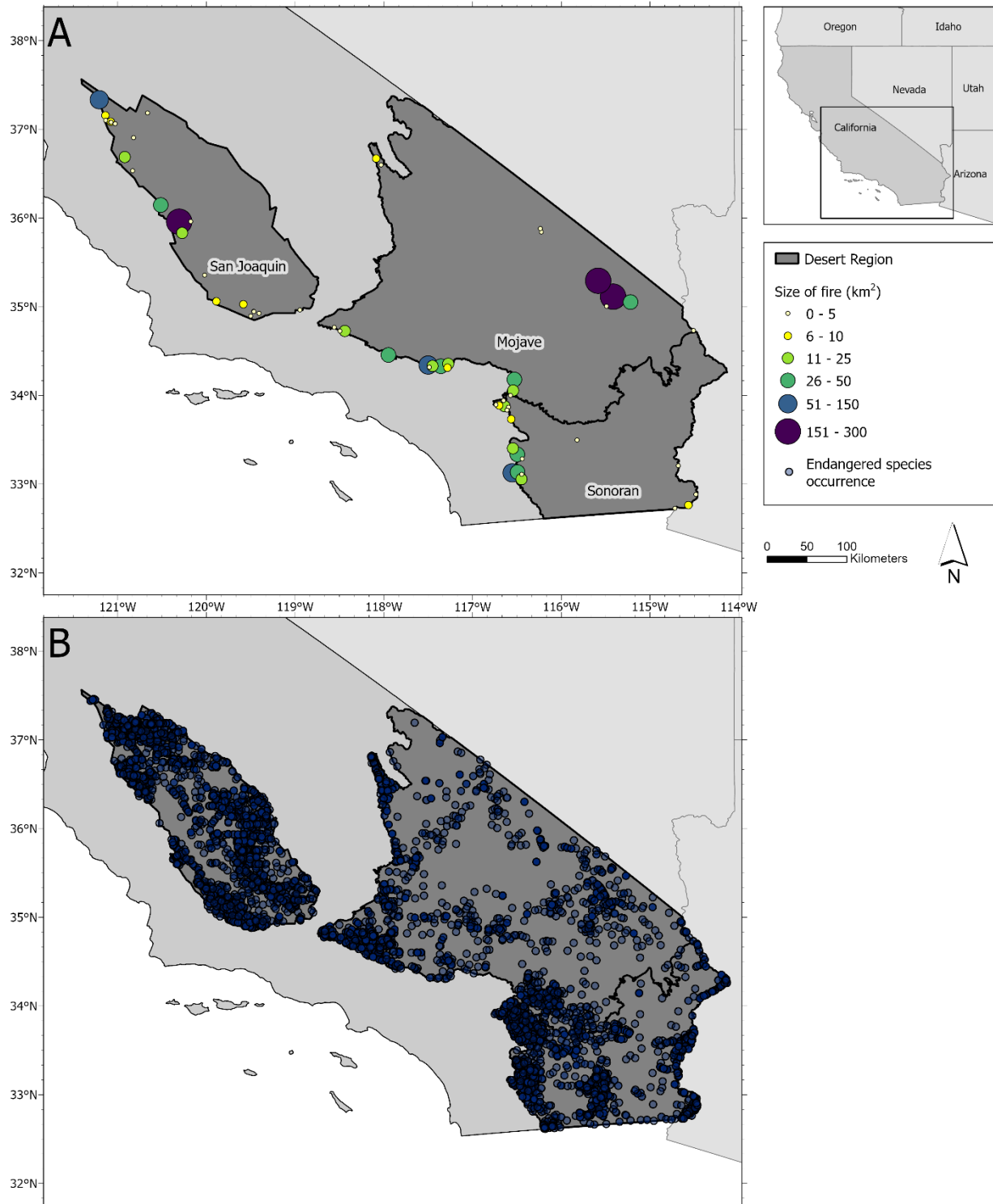
Figure 1: Map of southwest desert regions. The dark gray area surrounded by a black line indicates the desert boundary. (A) The location and relative size of fires (n=65) burned between 2000 and 2020 used in this study are indicated by the size and color of dot. (B) The location of endangered animal species occurrences retrieved from GBIF is represented by a blue dot using transparency to highlight hot spots.
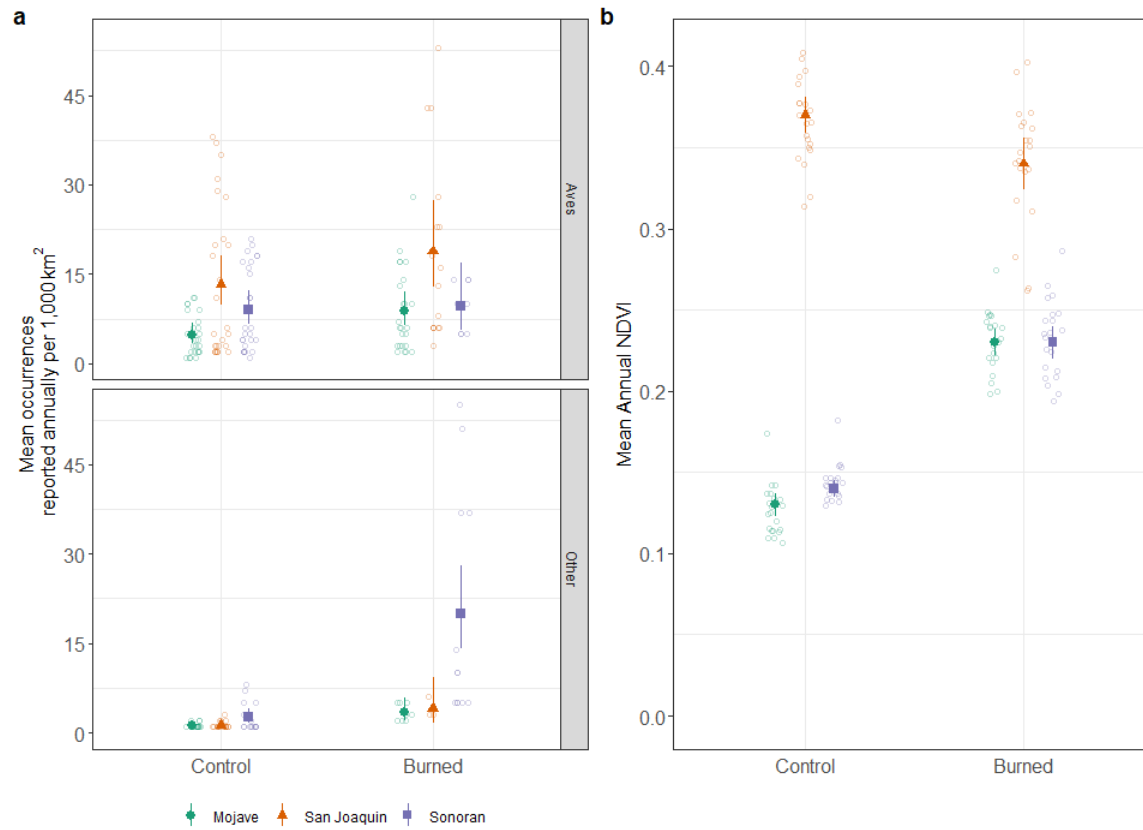
Figure 2: Comparison of (a) mean ES occurrences reported annually (1995-2020) per 1,000 km² and (b) mean annual (2000-2020) NDVI between burned and control group. Lighter open circles represent yearly average. Error bars show 95% confidence interval.
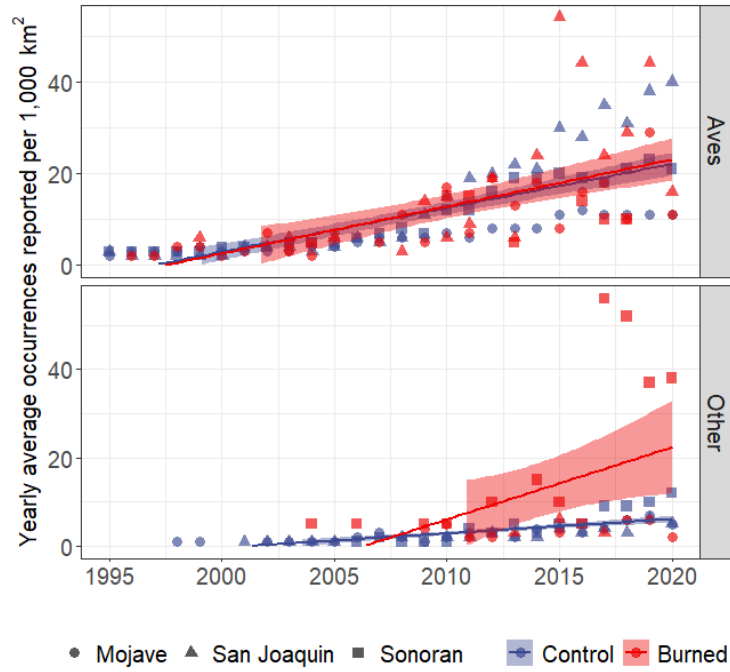
Figure 3: Yearly average of occurrences reported per 1,000 km^2 between 1995 and 2020. Top shows yearly averages for avian ES occurrences and the bottom shows non-avian ES occurrences.
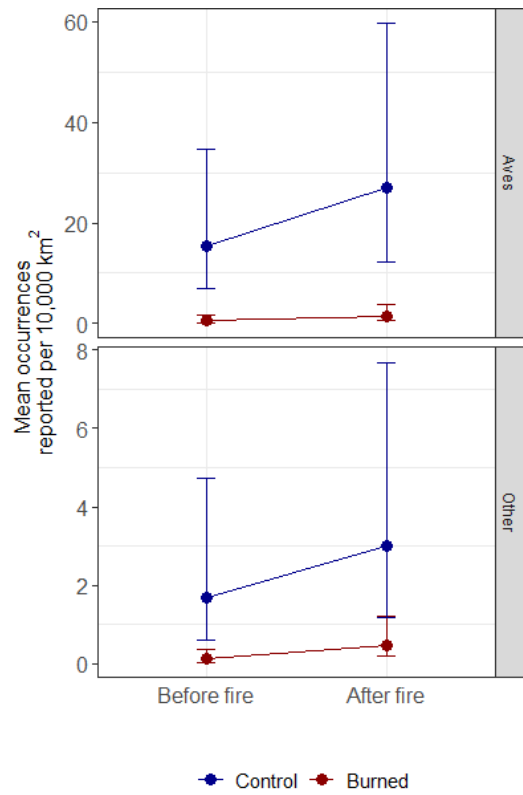
Figure 4: Before and after fire comparison of mean occurrences reported per 10,000 km^2 between burned and control sites.
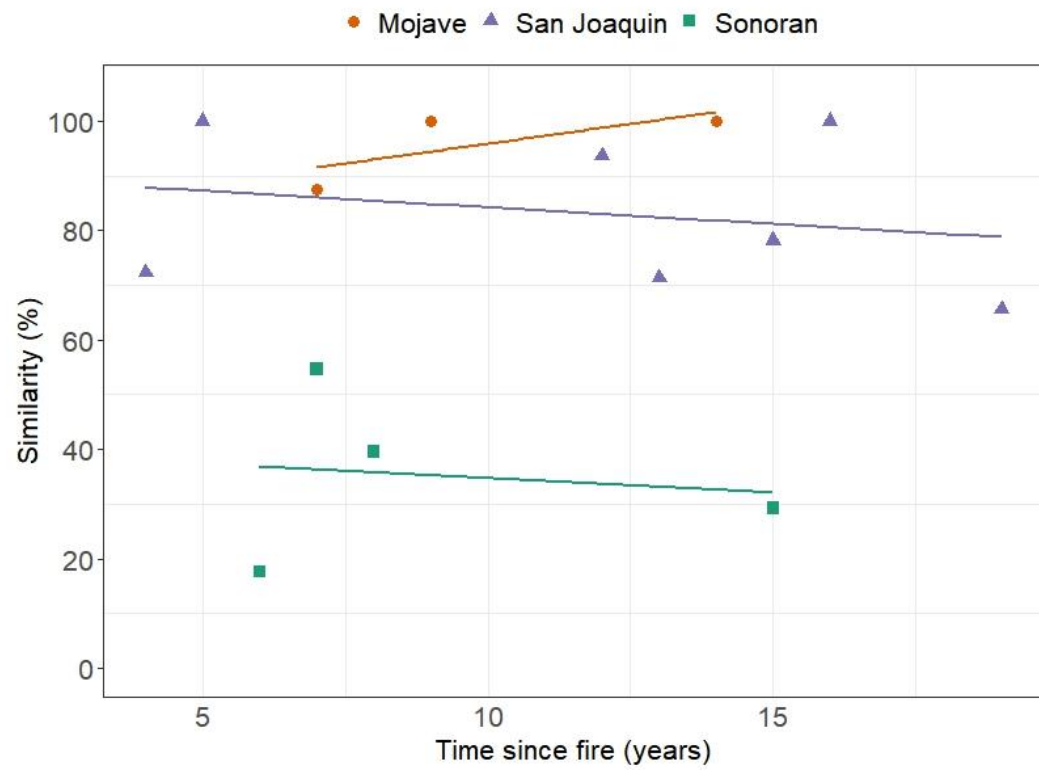
Figure 5: Incidence-based ChaoSørenson Similarity index for burned:unburned ES composition.
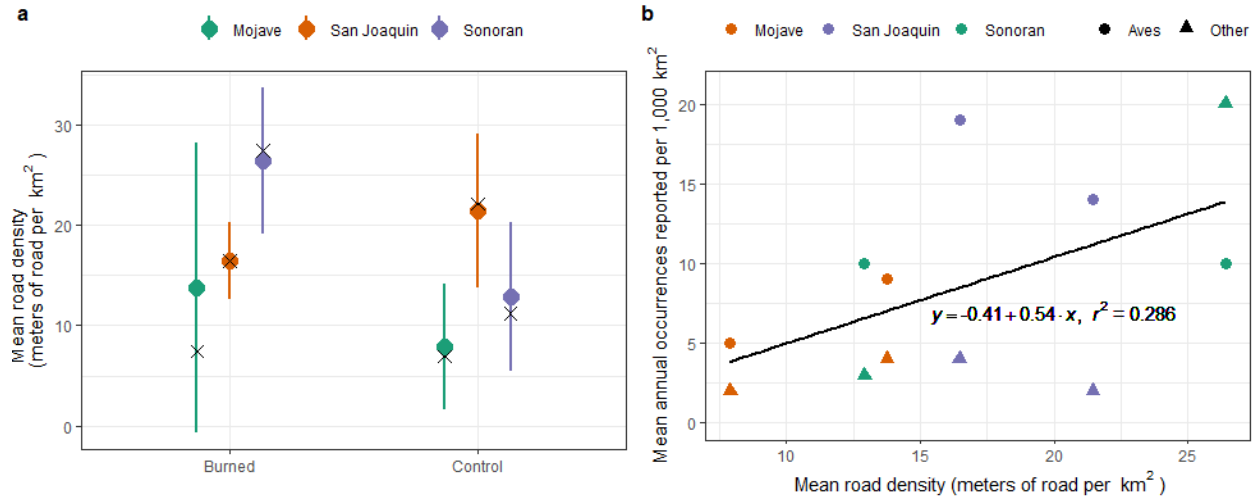
Figure 6: The influence of road density (m of road/km^2) on mean annual ES occurrences reported between 1995 and 2020. (A) Shows the mean road density across all three deserts. Median is represented by black X. Error bars represent standard deviation from the mean. (B) Models the relationship between mean road density and mean annual ES occurrences reported per 1,000 km^2 between 1995 and 2020.

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Appendices

Appendix A: Fires

Table 11: List of fires and area burned between 2000 and 2020 which contained reports of ES occurrence. 'NA' used where month of fire is not known.

Desert	Fire Name	Burn Date (Year)	Burn Data (Month)	Area burned within desert (km^2)
Mojave	BOBCAT	2020	September	45.4
	DOME	2020	August	178.9
	LAKE	2020	August	3.7
	MOFFAT	2018	April	5.1
	TECOPA	2017	September	0.1
	BLUE CUT	2016	August	72.5
	PILOT	2016	August	5.4
	TOPOCK	2016	April	1.4
	POWERHOUSE	2013	May	16.3
	RIVER	2013	February	1.6
	RANGE	2011	NA	1.0
	TECOPA	2010	August	0.4
	LOS FLORES	2007	March	16.6
	BLAIR	2006	NA	0.3
	SAWTOOTH COMPLEX	2006	July	44.6
	HACKBERRY COMPLEX	2005	June	258.1
	PARADISE	2005	June	12.0
	OLD	2003	October	41.5
	LOUISIANA	2002	June	2.8
	BLUE CUT	2002	June	13.8
San Joaquin	DEL PUERTO	2020	August	67.7
	MINERAL	2020	July	48.3
	HUDSON	2017	July	4.8

	GARZA	2017	July	197.9
	SANDY	2016	July	0.3
	SODA	2016	June	8.1
	MCCABE	2015	July	6.2
	SCHOOL	2014	April	0.9
	TURKEY	2012	July	10.2
	ROMERO	2011	July	1.8
	COTTON	2010	May	8.2
	PARKWAY	2009	June	1.7
	AVENAL	2008	August	3.8
	ROMERO	2008	July	3.1
	COTTONWOOD2	2008	July	1.9
	BROWN	2008	June	15.3
	SAN LUIS	2005	NA	1.3
	SODA	2005	June	1.8
	DOS AMIGOS	2005	June	3.0
	HWY166	2004	NA	0.3
	BASALT	2004	May	1.7
	NEW IDRIA	2003	May	2.6
	BECK	2000	NA	1.6
	ROMERO	2000	August	6.9
Sonoran	DRAPER	2017	September	3.0
	ALAMO	2016	July	0.6
	TRAM	2013	August	0.2
	MOUNTAIN	2013	July	7.8
	SENATOR	2012	NA	0.2
	STEWART	2012	August	43.0
	BANNER FIRE	2012	May	14.2
	EAGLE	2011	July	39.3

WINDY POINT	2011	September	2.2
NARROWS	2010	June	0.2
HAUGHTELIN	2010	May	9.4
INDIAN 2	2005	September	0.8
BLAISDELL	2005	August	17.0
ELM	2005	June	1.1
VERBENIA	2004	July	9.1
COYOTE	2003	July	12.4
PALM#1	2002	NA	0.4
PINES	2002	July	58.1