

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

Marina Goldgisser

A thesis submitted to the Faculty of Graduate Studies in partial fulfillment of the
requirements for the degree of Master of Science

Graduate program in Geography
YORK UNIVERSITY
TORONTO, ONTARIO

September 2023

© Marina Goldgisser, 2023

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 ChaoSørensen similarity index. Burned regions had higher vegetation productivity than unburned regions in some, but not all, deserts. Post-fire habitats continued to support ES, even 19 years after a fire. These 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

Acknowledgements

Table of Contents

Abstract.....	i
Acknowledgements.....	ii
Table of Contents.....	iii
List of Tables.....	v
List of Figures	vii
List of Abbreviated Terms and Acronyms.....	viii
Introduction	1
Ecological background	1
The value of citizen science in ecological research	3
Research Objectives	5
Methods.....	6
Study area.....	6
Endangered species.....	7
NDVI data.....	9
Data analysis.....	10
Results.....	13
Differences in mean annual NDVI.....	13
Differences in ES occurrences reporting.....	13
Impact of fires on ES occurrences	14
Recovery of ES post-fire using community composition measures	14
Discussion.....	15

Key findings.....	15
Vegetation	16
Fire impact on ES biodiversity	17
The cost of citizen science.....	21
Implications	24
Conclusion	26
Tables and Figures	27
References.....	44
Appendices	66
Appendix A: Fires	66
Appendix B: Supplementary material evaluating the influence of road density on mean annual ES occurrences reported between 1995-2020.	69
Methods.....	69
Results.....	69

List of Tables

Table 1: Description of desert bioregions in southwest California.-----	27
Table 2: List of endangered animal species included in occurrence data acquired from GBIF.	28
Table 3: List of datasets involved in GBIF report. All, except the Avian Knowledge Network, are datasets compiled by citizen science.-----	32
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. 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.-----	33
Table 5: Analysis of trend in yearly average reported ES occurrences from generalized linear mixed model (GLMM) for 1995-2020. All significant p -values ($p < 0.05$) are indicated in bold. -----	34
Table 6: Analysis of mean annual ES occurrences reported from generalized linear model (GLM) for 1995-2020. All significant p -values ($p < 0.05$) are indicated in bold. -----	35
Table 7: Results from emmeans post hoc test. Main effects, (top) desert region and (bottom) 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.	36
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. -----	37
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.-----	38

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.-----	38
Table 11: List of fires and area burned between 2000 and 2020 which contained reports of ES occurrence.-----	66

List of Figures

- Figure 1: Map of southwest desert regions. The dark gray area surrounded by a black line indicates the desert perimeter, and the dots indicate the location and relative size of fires ($n = 65$) burned between 2000 and 2020 used in this study. ----- 39
- 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. ----- 40
- Figure 3: Yearly average of occurrences reported per 1,000 km² between 1995 and 2020. Top shows yearly averages for avian ES occurrences and the bottom shows non-avian ES occurrences. ----- 41
- Figure 4: Before and after fire comparison of mean occurrences reported per 10,000 km² between burned and control sites. ----- 42
- Figure 5: Incidence-based ChaoSørensen Similarity index for burned:unburned ES composition. ----- 43
- Figure 6: Relationship between mean road density and mean annual ES occurrences reported per 1,000 km² between 1995 and 2020.----- 69

List of Abbreviated Terms and Acronyms

ES	Endangered animal species
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 (M. L. Brooks & Matchett, 2006; M. L. 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 (M. L. 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 Wagtenonk & 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; M. L. Brooks & Matchett, 2006; Minnich, 2018). Nevertheless, variation in precipitation and vegetation composition throughout the southwest desert systems influence fire frequency and severity. This in turn likely influences patterns in regional biodiversity (Pastor et al., 2011; Diffendorfer et al., 2012; Dorph et al., 2020).

All 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). 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). 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, an increase in the occurrence and spread of wildfire (M. L. 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; M. L. Brooks et al., 2004; M. L. 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.

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 (M. L. 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 establish (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; M. L. 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 of increasing support to ecological research and can help scientists meet the challenges of timely monitoring and assessment of impacted biodiversity following a large-scale environmental disturbance—like wildfires (Lawson et al., 2015; Kirchhoff et al., 2021). Data collected by citizen scientists (i.e., non-professionals) can be useful in modeling species distributions (Renner et al., 2015), predicting species richness (Farwell et al., 2021; Carroll et al., 2022) and monitoring population trends (Neate-Clegg et al., 2020). By crowd-sourcing data collection, land managers and researchers can better monitor ecosystem change and have 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 (McKinley et al., 2017). Increasing fire activity throughout southwestern North American deserts in recent decades (M. L. Brooks &

Matchett, 2006; Dennison et al., 2014; Syphard et al., 2017) will require land managers and conservationists to efficiently and effectively monitor biological change. Traditional ground survey work can be powerful but at times also limited in scope, spatially and temporally. More than 30% of the total land area in California is defined as a desert ecoregion (Syphard & Keeley, 2020). Fires can be large, unpredictable, and multiple burns can occur in a single season (M. L. Brooks & Matchett, 2006). Furthermore, 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). Citizen science and publicly accessible data through a database like the Global Biodiversity Inventory Facility (GBIF) 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). GBIF aggregates data from a variety of sources, including eBird and iNaturalist. These sources typically engage citizen scientists in wildlife observation and data collection. Projects, such as the California Native Plant Society's Fire Follower—which encourages participants to record the response of plants in areas burned following the 2020 and 2021 fire season (<https://www.cnps.org/fire-followers>) – can provide valuable information in places and at scales that may not always be possible for researchers to access. Well-planned citizen science initiatives lend an opportunity for increased spatial and temporal coverage marked with fine-scale observations. For instance, following the 2019-2020 unprecedented Australian bushfire season, Kirchhoff and colleagues (2021) were able to mobilize over 200 volunteers through iNaturalist, an app where participants can submit observations of any living or once-living organism, to assess the severity and impact of fires on the over 50 million hectares of land. Such large-scale monitoring 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.

Understanding the impacts of regional environmental change, such as changes in wildfire regime, necessitates distinguishing whether population trends and shifts in community composition result from natural temporal turnover or from a more dramatic response to direct disturbance impacts. Long-term datasets are critical for examining temporal patterns of population dynamics and assessing the impact of disturbances on biodiversity (Magurran et al., 2010). On average, citizen science projects run 10 years or longer (Theobald et al., 2015), but require continued support and public engagement to maintain successful long-term monitoring (Bayraktarov et al., 2019). Long-running citizen science projects can collect data pre- and post-intervention (or disturbance) providing important information on community response to change (Newman et al., 2017). Incorporating citizen science projects in research, and designing them to be sustainable and maintainable, can improve spatial and temporal coverage, lead to a better understanding of macro-regional ecosystem functions, and provide a more accurate assessment of vulnerable ecosystems.

Research Objectives

This study used open-source biodiversity data from GBIF 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 according to fire activity. Secondly, 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’ or ‘never-burned’ 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 have these trends been impacted by the presence of fire?

- (2) Has the occurrence of fire impacted the number of ES occurrences reported?
- (3) Is there a long-term (20-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 to never-burned areas. After assessing trends in ES occurrences reported over a 25-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 reported ES. Finally, I modeled 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 km²) of the total land area in California (Figure 1, Table 1). More than half of this amount is accounted for by the MOJ alone. Desert bioregions are characterized by their low precipitation, arid soils, sparse vegetation, and desert-adapted plants and vertebrates (Whittaker, 1975; Bailey, 2009; M. L. 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 (M. L. 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 (M. L. Brooks et al., 2018). Desert shrubland comprises much of the SON and MOJ—dominated by creosote bush scrub and saltbush scrub (M. L. 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 shapefiles for the MOJ and SON ecoregions were retrieved from the United States Environmental Protection Agency (EPA) data repository (<https://www.epa.gov/eco-research/ecoregions>). The SJD shapefile was provided by The Nature Conservancy (Lortie et al., 2018). The SJD describes the western and southern two-thirds of the San Joaquin Valley as well as the adjacent Carrizo Plain and Cuyama Valley because of its 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/>). Only fires that burned within the three desert areas between 2000 and 2020 and contained reports of endangered animal species occurrence were selected for use in the study. Fire boundaries were clipped to only include the area within desert ecoregions. Though some fires spread beyond desert ecoregion, only the area burned within the defined deserts were used in this study. 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 km² to 258 km², with a median size of 4.3 km² and a mean size of 21.6 km² 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.

The total of recorded occurrences used from GBIF was 24,936 animal records for the 36 unique ES (Table 2) within the three desert systems from 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.

Almost 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 nearly 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. Animals were categorized into 11 functional classes: birds, large mammals, small mammals, tortoises, frogs & toads, lizards & geckos, salamanders, butterflies & moths, bees, beetles, or aquatic. Birds were further grouped into 4 functional groups: passerines, raptors, woodpeckers, or rails & geese.

NDVI data

An aggregate mean normalized difference vegetation index (NDVI) was used as a proxy for vegetation structure to evaluate the relationship between vegetation quality, fire activity, and occurrences reported over a 20-year period (2000-2020). NDVI values range from ≤ 0 (no photosynthetic activity) to +1 (high photosynthetic activity) and are derived from the ratio

$$\text{NDVI} = \frac{\text{NIR} - \text{R}}{\text{NIR} + \text{R}} \quad (\text{Equation 1})$$

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/dataproduct/mod13.php>) 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 unburned sites and burned sites in each desert ecoregion separately.

Data analysis

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-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 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). Time (year) and treatment groups were included as an explanatory variables. 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 functional group. The models were fit using a negative binomial distribution to account for overdispersion in the data (Stoklosa et al., 2022). ES occurrences are scaled to the number reported per 1,000 km² 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.

Fire-related impact was further evaluated using before/after comparisons and analyzed using generalized linear mixed models (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 area of 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 to include only non-avian species occurrences as well as only bird 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 km² 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

$$\text{ChaoSørensen}_{\text{incidence}} = \frac{2UV}{U+V} \quad (\text{Equation 2})$$

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 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 20-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; Lüdtke 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

The mean annual NDVI (Figure 2; Table 4) was significantly different between treatment groups (ANOVA¹, $p < 0.001$) and desert ecoregions (ANOVA¹, $p < 0.001$). Post hoc comparison shows that mean annual NDVI was greater in SJD than both the MOJ (Tukey HSD, difference = 0.175, $p < 0.001$) and the SON (Tukey HSD, difference = 0.165, $p < 0.001$). However, there was no significant difference of mean annual NDVI between MOJ:SON burned treatments (Tukey HSD, difference = 0.002, $p = 0.819$), but the SON had a slightly higher mean annual NDVI in the control treatment than the MOJ (Tukey HSD, difference = 0.017, $p < 0.001$).

Separate ANOVA results were estimated for SJD, MOJ and SON (inclusive of both treatments)², burned treatment only of MOJ and SON, and control treatment only of MOJ and SON. 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 (ANOVA², $p < 0.001$; Tukey HSD, difference of 0.097, $p < 0.001$).

Differences in ES occurrences reporting

Over the 25-year period (1995-2000), both treatment groups showed an overall positive trend in the yearly average ES occurrences reported for all three deserts (Figure 3; Table 5); however, there was a greater positive trend in the annual non-avian ES occurrences reported in

¹Does not meet normality assumption

² Does not meet normality assumption

the burned treatment than in the control treatment (post-hoc contrast, $p < 0.001$). 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*).

More ES occurrences per 1,000 km² 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, $p < 0.001$).

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, whether there was a fire disturbance or not, reports of ES occurrences increased in future years. There were a significantly greater number of ES occurrences reported in the control group than in the burned group (post hoc contrast, $p < 0.001$). To explore sensitivity to the most reported class, I conducted a second set of GLMMs that reanalyzed the ES occurrences reported pre- and post-fire excluding aves. 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 in burned:unburned similarity of ES composition across the three desert systems (Figure 5; Table 9; LM, $p = 0.512$), suggesting that the community was

composed of similar endangered species that continued to occupy the regions regardless of fire effects. Among the three deserts, the SON had the lowest similarity between burned:unburned ES composition ($30 \pm 10\%$ [ChaoSørensen similarity mean \pm standard error], post-hoc contrast, $p = 0.001$). Although the MOJ had the greatest similarity between burned:unburned ES composition ($95 \pm 10\%$ [ChaoSørensen similarity mean \pm standard error]), 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 (ES) in California desert systems and I found that fire was a significant effect, but not necessarily a negative one, for the 36 ES that were reported within these regions. Pre- and post-fire comparisons suggest that non-avian ES were potentially more impacted by fire than avian ES. Greater reporting of non-avian ES compared to avian ES might be explained by the highly mobile nature of birds and their ability to disperse farther to find resources and meet habitat requirements (Martin & Fahrig, 2018). 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 productivity. There were no significant shifts in ES composition between burned and unburned desert sites in the time following a fire. This suggests that burned habitats are still able to support ES communities and that ES continue to visit or occupy burned desert areas. The findings here

support the use of GBIF data as a tool to explore and monitor landscape-level analysis of ES responsiveness to fire.

Vegetation

In this study, burned sites in the MOJ and SON exhibited greater plant productivity than control sites. The SJD, however, had the highest mean annual NDVI among the three deserts, but mean annual NDVI did not differ between burn and control treatment groups. Most fires in the MOJ and SON occurred on the western edge of the desert ecoregions and at higher elevations (M. L. 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 (M. L. 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). 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). The SJD, once characterized by saltbush scrub communities mixed with patches of native grasses, is now largely dominated by non-native grasslands (Kelly et al., 2006; Wills, 2018). 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 creates a continuous fuel bed in the typically barren desert landscapes. Moreover, previous studies suggest that 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% lower 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). Thus, not only does vegetation have the ability to influence

desert fire regimes, but fire regimes also have the ability to influence desert plant composition. This underscores the need to expand upon the tools available to monitor landscape-level biodiversity changes in the wake of fire disturbance—especially as current climate models predict increased desert fires activity in the future (Abatzoglou & Kolden, 2011).

Fire impact on ES biodiversity

Endangered animal communities continued 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 and modeling the number of ES reported in burned:unburned areas throughout the MOJ, SON, and SJD. The increase in ES occurrences reported post-fire than pre-fire in both treatment groups suggests that fires do not appear to have a 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—possibly because of fire-mediated habitat change (Abella et al., 2009; Clapp & Beck, 2016; Dorph et al., 2020; Geary et al., 2020). Abella and colleagues (2009) suggests 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. Thus, fire disturbance can improve habitat for desert species that favor open spaces (Monasmith et al., 2010; Horn et al., 2012). However, 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). Despite loss in native shrub diversity, burned areas can recover shrub coverage and density in the years following fire (Steers & Allen, 2011; Horn et al., 2015). The varying influence of burns on desert habitats can explain the overall resilience of ES to fire disturbance.

Upstream and upland fires can improve aquatic habitats for aquatic and amphibian ES 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 and colleagues (2010) has shown Yuma Ridgway's rails (*Rallus obsoletus yumanensis*) can also respond positively to cleared dense vegetation in burned wetland habitat; however, this is not supported in this study (i.e. 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). The ability of fire to impact riparian and aquatic habitats directly and indirectly has important implications for the at-risk species that inhabit these ecological zones in deserts.

Fire-mediated habitat change can also 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 predators may also be attracted to recently burned areas because of increased prey availability—meanwhile, simultaneously benefitting from reduced habitat cover that 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. Future research on the direct and indirect effects of fire on predator-prey interactions in desert systems can aid in understanding how specific predator-prey interactions may shift in response to changing fire regimes across California's desert regions.

Not all fire-impacted desert lands remain as the open sparsely-vegetated shrublands preferred by many desert-adapted species. This study did not compare vegetative communities

pre- and post-fire; however, 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—changing the vegetative composition of desert ecosystems (St. Clair et al., 2016; Bishop et al., 2020). Moreover, 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. 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 nitratooides nitratooides*), San Joaquin antelope squirrels (*Ammospermophilus nelsoni*), and Mohave ground squirrels (*Xerospermophilus mohavensis*). The SJD is an example of a desert proposed to have once be dominated by desert shrubland able to support desert-adapted plants and animals, but the invasion of non-native grasses has created an unrecognizable landscape—either dominated by dense grassland or shrubland with a dense non-native grass understory (Germano et al., 2011, 2012). 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, further research would provide a deeper understanding on species-specific responses to fire-mediated habitat change.

It is clear that fire-ES dynamics are complex. While quadrupedal rodents prefer shrub cover—which may be consumed by fire (Sharp Bowman et al., 2017), bipedal rodents, such as kangaroo rats (*Dipodomys* spp.), benefit from the increased open space created by fires and, simultaneously, help limit the establishment of invasive grasses through seed predation (Horn et al., 2012; Sharp Bowman et al., 2017; Bishop et al., 2020; Chock 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.

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). Therefore, ES response to fire might display a temporal lag as species shift away from burned areas and then shift-back during recovery. This implies recovery and reuse of habitat by ES post-fire is viable and that arid lands managed by fire can still support ES.

There are multiple mechanisms to explain this pattern in ES persistence following fire disturbance. 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/shade following a fire (Drake et al., 2015; Germano, 2019). In the absence of shrubs, blunt-nosed leopard lizards may still be present; however, they must 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 and 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 likely utilize burned and unburned patches to meet their resource demands (Mendelsohn et al., 2008). These results are promising for managers seeking to protect at-risk species in burned areas.

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, therefore, has been most resistant to invasive grasses (M. L. 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 arrived first to—and more easily established in—the more mesic regions of the SJD and MOJ, giving those ecosystems more time to acclimate (M. L. 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 (M. L. 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 cost of citizen science

Citizen science can play a key role in assessing and monitoring ecosystem-wide change that occurs following a fire disturbance. Currently, it is 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 number of biodiversity-focused citizen science projects have greatly increased in the last several decades (Theobald et al., 2015), and changes in policies and grant requirements 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 (Loarie, 2020; Team eBird, 2021). The gradual increase of occurrences reported in all three desert regions and across both burned and unburned sites within the last 20 years reflects the growing use of citizen science data collection. GBIF is an omnium gatherum of biodiversity datasets, nonetheless citizen science data submissions—specifically observations submitted through eBird and iNaturalist—make up a majority of species occurrence records stored on GBIF. eBird (launched in 2002) and iNaturalist (launched in 2008) allow for non-structured data collection, which is made easier by apps that allow users to submit their observations from their smartphone or computer. 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 (Appendix B: Supplementary material); 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). Although some apps, like eBird, allow for observers to submit complete checklists (i.e., lists that contain every species observers were able to identify to their best effort) that can be used to generate presence-absence data, 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 and colleagues (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 rare species to account for the effect of unseen (i.e., unreported) shared species between two communities (Chao et al., 2005). Additionally, combining citizen science data sets with professional data sets or ground-truthing results can improve the capacity of citizen science data to monitor environmental change (Crall et al., 2015). 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).

Implications

Three recommendations are made for land managers to reduce the threat to ES following fire disturbance: (1) focus on maintaining open shrub habitat, (2) mediate the impact of fire by providing temporary reprieve from fire-mediated habitat changes, and (3) incorporate citizen science data into biodiversity monitoring.

Vegetation composition influences patterns of animal community recovery post-disturbance and, therefore, land managers should focus on managing invasive species and promoting native habitat in the context of fire (Diffendorfer et al., 2012). Fires can increase the potential for invasive grasses to establish in open desert shrub habitat (D'Antonio & Vitousek, 1992; Hunter et al., 2006). However, not all fires promote the invasion on non-native grasses. In fact, some land managers use frequent prescribed burns to clear denser vegetation, control invasive grass species, and promote suitable open habitat for at-risk species (M. L. Brooks & Pyke, 2001; Germano et al., 2001; Salvatori et al., 2001; Conway et al., 2010; Green et al., 2015; Clapp & Beck, 2016). The probability of non-native grass invasion in burned landscapes is influenced by fire severity and viable seed source from nearby unburned patches (Hunter et al., 2006; St. Clair & Bishop, 2019). Therefore, managers need to be careful when selecting areas for prescribed burns to reduce the potential for plant invasion. 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 burns.

Impacts of unpredictable fires that cause landscape-wide habitat change can be alleviated to support ES recovery short-term. Desert animals continue to use burned habitat structures for thermoregulation (Drake et al., 2015; Ivey et al., 2020); therefore, managers can deploy artificial shelters in burned areas to provide essential microhabitats while native vegetation recovers—especially in areas where plant structures were completely consumed by fire (Ghazian et al., 2020). Additionally, land managers can promote the recovery of native vegetation and further provide habitat structures by introducing native foundation plants into

disturbed communities (Filazzola & Lortie, 2014; Lortie et al., 2022). Ameliorating the effects of fire disturbance in desert habitats can help promote the short-term and long-term survival of ES species.

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 can 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 and colleagues (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. Citizen science can be a valuable tool in providing information relevant for management, especially in the face of a rapidly changing environment.

Conclusion

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 (M. L. Brooks et al., 2004; M. L. Brooks & Zouhar, 2008; Balch et al., 2013; Underwood et al., 2019). This work showed that citizen science data will often be a critical tool to support landscape-level ecological research. Open-source data—collected mostly by citizen scientists—from GBIF was used to understand the impacts of fires in the 21st century on endangered species in the Mojave, San Joaquin, and Sonoran desert regions in this study. The general trend of increasing endangered species occurrences over the last 25 years across all regions is to be expected, particularly with the introduction and increased adoption of citizen science reporting apps by the general public. The higher number of ES, particularly non-avian ES, 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; M. L. Brooks & Esque, 2002). Although this offers hope for land managers responsible for ES protection in desert lands, potential for observer bias to visit sites that are likely to have animal encounters and over-report rare species should encourage managers to strengthen data with ground-truthing when possible. The expansive scale of desert landscapes and potential for increased fire frequency and severity in the future should encourage land managers to continue using citizen science data to support monitoring efforts of ES biodiversity in fire-disturbed areas.

Tables and Figures

Table 1: Description of desert bioregions in southwest California.

Desert region^a	Total Area (km²)	Mean annual precipitation at valley floors (mm)^b	Total area burned (km²) 2000-2020	Total Occurrences Reported per 1000 km² 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 included in occurrence data acquired from GBIF.

Species	Common Name	Federal Status	State Status	Total number of occurrences reported within desert region						
				Mojave		San Joaquin		Sonoran		Total
				Burned	Control	Burned	Control	Burned	Control	
<i>Agelaius tricolor</i>	Tricolored blackbird		ST	13	1275	24	2840	1	317	4470
<i>Ambystoma californiense</i>	California tiger salamander	FE	ST				1			1
<i>Ammospermophilus nelsoni</i>	Nelson's antelope squirrel		ST			3	107			110
<i>Anaxyrus californicus</i>	Arroyo toad	FE		2						2
<i>Batrachoseps stebbinsi</i>	Tehachapi slender salamander		ST		1					1
<i>Bombus crotchii</i>	Crotch bumble bee		SC		6		6	1	3	16
<i>Branta hutchinsii</i>	Cackling goose	FDR					4			4
<i>Buteo swainsoni</i>	Swainson's hawk		ST	13	1158	26	4094	4	1114	6409
<i>Colaptes chrysoides</i>	Gilded northern flicker		SE	74	86				27	187
										28

<i>Coleonyx switaki</i>	Barefoot gecko		ST						3	3
<i>Cyprinodon macularius</i>	Desert pupfish	FE	SE					8	13	21
<i>Danaus plexippus</i>	Monarch		FC	3	32	1	18		50	104
<i>Dinacoma caseyi</i>	Casey's June beetle	FE			1				3	4
<i>Dipodomys ingens</i>	Giant kangaroo rat	FE	SE				17			17
<i>Dipodomys nitratoides</i>	Tipton kangaroo rat	FE	SE				2			2
<i>Empidonax traillii</i>	Willow flycatcher		SE	36	4047	10	453	6	1092	5644
<i>Euproserpinus euterpe</i>	Kern primrose sphinx moth		FT				1			1
<i>Falco peregrinus</i>	American peregrine falcon	FDR	SDR						1	1
<i>Gambelia sila</i>	Blunt-nosed leopard lizard	FE	SE				49			49
<i>Gopherus agassizii</i>	Desert tortoise	FT	ST ¹	8	412			1	127	548
<i>Gymnogyps californianus</i>	California condor	FE	SE		8	5	442			455

<i>Haliaeetus leucocephalus</i>	Bald eagle	FDR	SE	3	117	50	815		166	1151
<i>Melanerpes uropygialis</i>	Gila woodpecker		SE		6			4	2849	2859
<i>Micrathene whitneyi</i>	Elf owl		SE						3	3
<i>Ovis canadensis</i>	Peninsular bighorn sheep	FE	ST	2	113			23	433	571
<i>Pelecanus occidentalis</i>	California brown pelican	FDR	SDR				1		10	11
<i>Puma concolor</i>	Mountain lion		SC	1	2			13	32	48
<i>Rallus obsoletus</i>	Yuma Ridgway's rail	FE	ST					1	21	22
<i>Rana boylei</i>	Foothill yellow-legged frog		SE				3			3
<i>Rana draytonii</i>	California red-legged frog	FT			1		1			2
<i>Rana muscosa</i>	Southern mountain yellow-legged frog	FE	SE		1					1
<i>Riparia riparia</i>	Bank swallow		ST	5	1403	1	149		483	2041
<i>Uma inornata</i>	Coachella Valley fringe-toed lizard	FT	SE		15			5	124	144

<i>Vireo bellii</i>	Least Bell's vireo	FE	SE	5				3	8	
<i>Vulpes macrotis</i>	San Joaquin kit fox	FE	ST			21			21	
<i>Xerospermophilus mohavensis</i>	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 reported
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. 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 <i>SJD</i>	Treatment group	1	0.006	0.006	6.099	0.018
NDVI [†] <i>SON + MOJ</i>	Treatment group	1	0.196	.0196	551.6	< 0.001
NDVI <i>Burned: SON + MOJ</i>	Desert region	1	<0.0001	<0.0001	0.053	0.819
NDVI <i>Control: SON + MOJ</i>	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 from generalized linear mixed model (GLMM) for 1995-2020. All significant p-values ($p < 0.05$) are indicated in bold.

		df	Chi-square	Pr(>Chi-square)
All ES	(Intercept)	1	72.75	<0.001
	Year	1	75.06	<0.001
	Treatment group	1	13.25	0.0003
Avian ES	(Intercept)	1	2.68×10^7	<0.001
	Year	1	-3.17×10^7	1.00
	Treatment group	1	3.00×10^{-4}	0.99
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 from generalized linear model (GLM) for 1995-2020. All significant p-values ($p < 0.05$) are indicated in bold.

		<i>df</i>	Deviance	Residual <i>df</i>	Residual deviance	<i>p</i> - 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. Main effects, (top) desert region and (bottom) 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.

	Desert Region	Response	SE	df	z ratio	p -value
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	Response	SE	df	z ratio	p -value
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.

		<i>df</i>	Chi-square	Pr(>Chi-square)
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.

		<i>df</i>	Sum Sq	Mean Sq	<i>F</i> value	<i>p</i> -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	<i>df</i>	<i>t</i> ratio	<i>p</i> -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

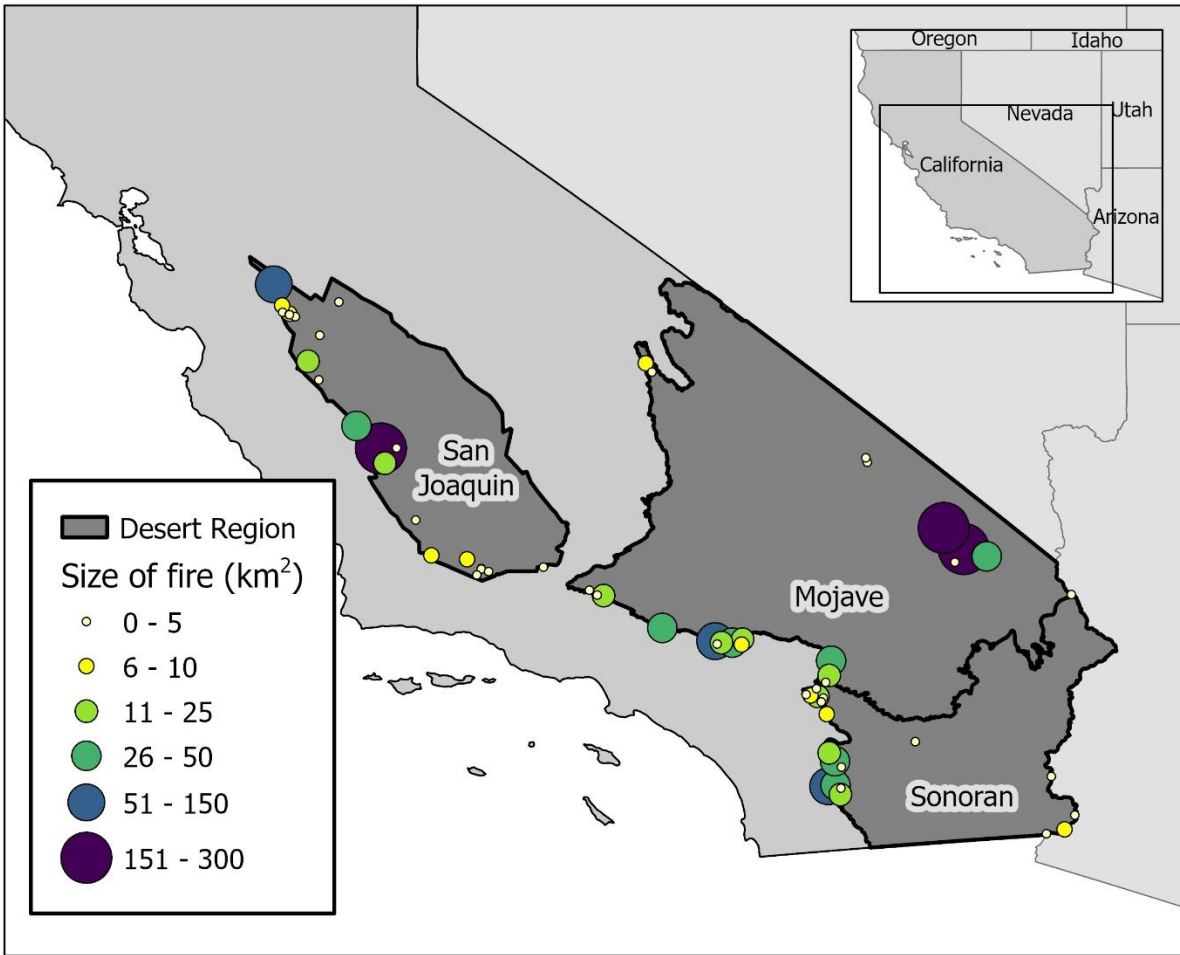


Figure 1: Map of southwest desert regions. The dark gray area surrounded by a black line indicates the desert perimeter, and the dots indicate the location and relative size of fires ($n = 65$) burned between 2000 and 2020 used in this study.

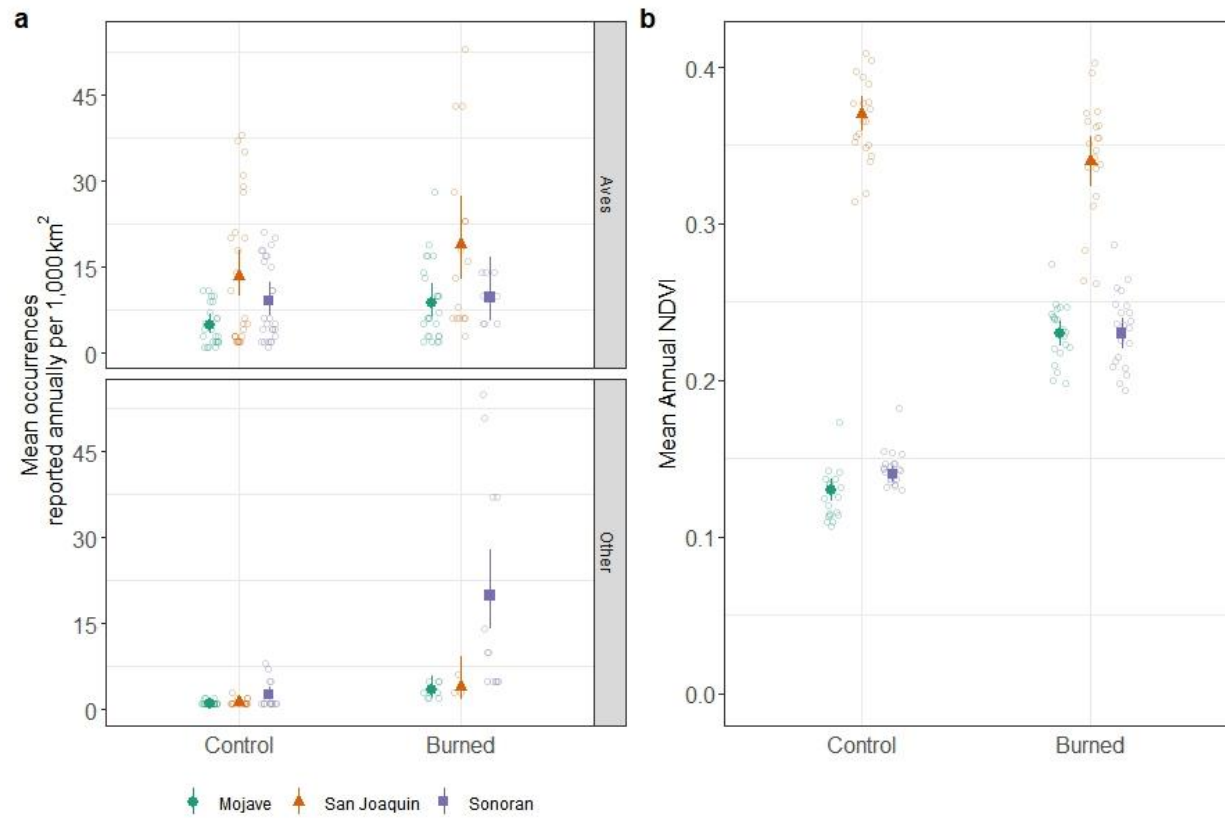


Figure 2: Comparison of (a) mean ES occurrences reported annually (1995-2020) per 1,000 km² and (b) mean annual (2000-2020) NDVI between burned and control group. Lighter open circles represent yearly average. Error bars show 95% confidence interval.

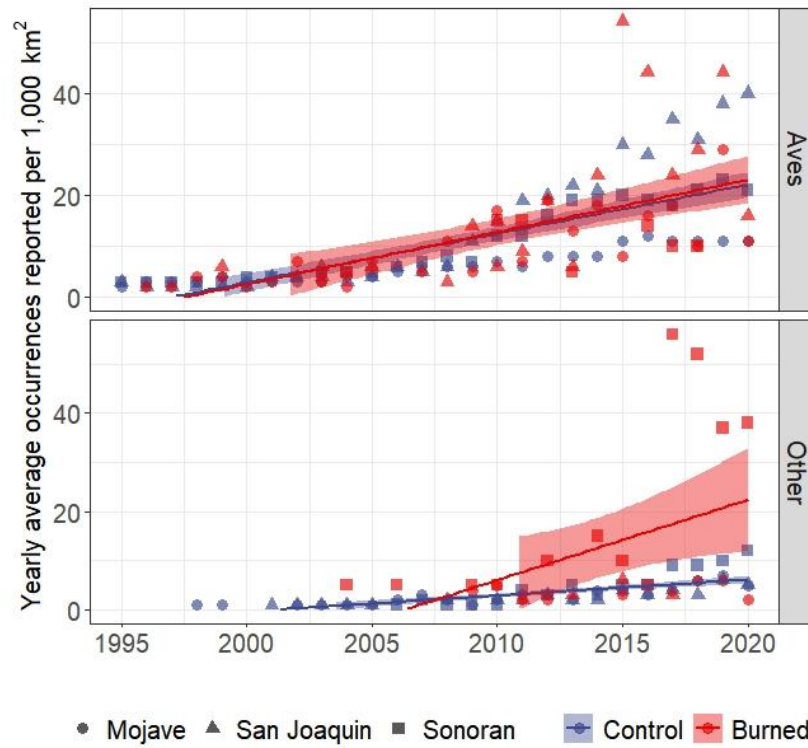


Figure 3: Yearly average of occurrences reported per 1,000 km² between 1995 and 2020. Top shows yearly averages for avian ES occurrences and the bottom shows non-avian ES occurrences.

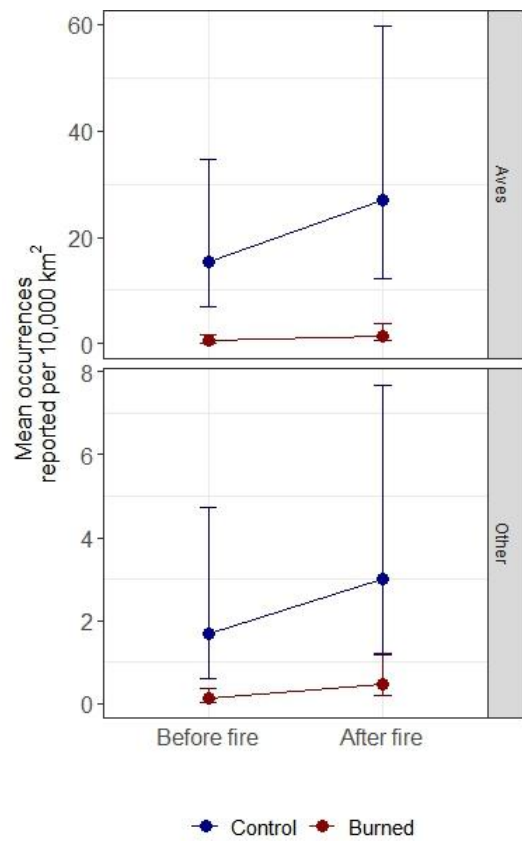


Figure 4: Before and after fire comparison of mean occurrences reported per 10,000 km² between burned and control sites.

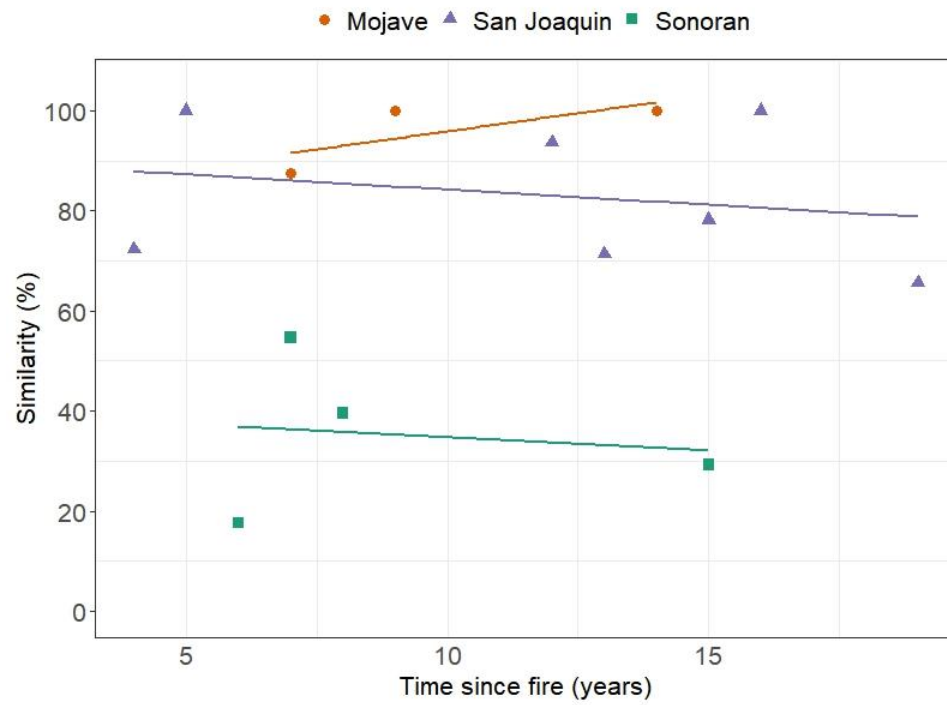


Figure 5: Incidence-based ChaoSørensen Similarity index for burned:unburned ES composition.

References

- Abatzoglou, J. T., & Kolden, C. A. (2011). Climate Change in Western US Deserts: Potential for Increased Wildfire and Invasive Annual Grasses. *Rangeland Ecology & Management*, 64(5), 471–478. <https://doi.org/10.2111/REM-D-09-00151.1>
- Abella, S. R. (2009). Post-fire plant recovery in the Mojave and Sonoran Deserts of western North America. *Journal of Arid Environments*, 73(8), 699–707. <https://doi.org/10.1016/j.jaridenv.2009.03.003>
- Abella, S. R., Engel, E. C., Lund, C. L., & Spencer, J. E. (2009). *Early post-fire plant establishment on a Mojave Desert burn*. 56, 13.
- Bailey, R. G. (2009). *Ecosystem Geography*. Springer New York. <https://doi.org/10.1007/978-0-387-89516-1>
- Balch, J. K., Bradley, B. A., D'Antonio, C. M., & Gómez-Dans, J. (2013). Introduced annual grass increases regional fire activity across the arid western USA (1980–2009). *Global Change Biology*, 19(1), 173–183. <https://doi.org/10.1111/gcb.12046>
- Barboza, F. R., & Defeo, O. (2015). Global diversity patterns in sandy beach macrofauna: A biogeographic analysis. *Scientific Reports*, 5(1), Article 1. <https://doi.org/10.1038/srep14515>
- Bayraktarov, E., Ehmke, G., O'Connor, J., Burns, E. L., Nguyen, H. A., McRae, L., Possingham, H. P., & Lindenmayer, D. B. (2019). Do Big Unstructured Biodiversity Data Mean More Knowledge? *Frontiers in Ecology and Evolution*, 6, 239. <https://doi.org/10.3389/fevo.2018.00239>
- Bird, T. J., Bates, A. E., Lefcheck, J. S., Hill, N. A., Thomson, R. J., Edgar, G. J., Stuart-Smith, R. D., Wotherspoon, S., Krkosek, M., Stuart-Smith, J. F., Pecl, G. T., Barrett, N., & Frusher, S. (2014). Statistical solutions for error and bias in global citizen science datasets. *Biological Conservation*, 173, 144–154. <https://doi.org/10.1016/j.biocon.2013.07.037>

- Bishop, T. B. B., Gill, R. A., McMillan, B. R., & St. Clair, S. B. (2020). Fire, rodent herbivory, and plant competition: Implications for invasion and altered fire regimes in the Mojave Desert. *Oecologia*, 192(1), 155–167. <https://doi.org/10.1007/s00442-019-04562-2>
- Boakes, E. H., McGowan, P. J. K., Fuller, R. A., Chang-qing, D., Clark, N. E., O'Connor, K., & Mace, G. M. (2010). Distorted Views of Biodiversity: Spatial and Temporal Bias in Species Occurrence Data. *PLoS Biology*, 8(6), e1000385. <https://doi.org/10.1371/journal.pbio.1000385>
- Bock, C. E., & Block, W. M. (2005). Fire and birds in the southeastern United States. *Studies in Avian Biology*, 30, 14–32.
- Bolker, B. M., Brooks, M. E., Clark, C. J., Geange, S. W., Poulsen, J. R., Stevens, M. H. H., & White, J.-S. S. (2009). Generalized linear mixed models: A practical guide for ecology and evolution. *Trends in Ecology & Evolution*, 24(3), 127–135. <https://doi.org/10.1016/j.tree.2008.10.008>
- Brooks, M. E., Kristensen, K., van Benthem, K. J., Magnusson, A., Berg, C., W., Nielsen, A., Skaug, H. J., Mächler, M., & Bolker, B. M. (2017). GlimmTMB Balances Speed and Flexibility Among Packages for Zero-inflated Generalized Linear Mixed Modeling. *The R Journal*, 9(2), 378. <https://doi.org/10.32614/RJ-2017-066>
- Brooks, M. L. (2012). *Effects of high fire frequency in creosote bush scrub vegetation of the Mojave Desert*. 8.
- Brooks, M. L., D'Antonio, C. M., Richardson, D. M., Grace, J. B., Keeley, J. E., DiTomaso, J. M., Hobbs, R. J., Pellant, M., & Pyke, D. (2004). Effects of Invasive Alien Plants on Fire Regimes. *BioScience*, 54(7), 677–688. [https://doi.org/10.1641/0006-3568\(2004\)054\[0677:EOIAP0\]2.0.CO;2](https://doi.org/10.1641/0006-3568(2004)054[0677:EOIAP0]2.0.CO;2)
- Brooks, M. L., & Esque, T. C. (2002). Alien Plants and Fire in Desert Tortoise (*Gopherus agassizii*) Habitat of the Mojave and Colorado Deserts. *Chelonian Conservation and Biology*, 4(2), 330–340.

- Brooks, M. L., & Matchett, J. R. (2006). Spatial and temporal patterns of wildfires in the Mojave Desert, 1980–2004. *Journal of Arid Environments*, 67, 148–164.
<https://doi.org/10.1016/j.jaridenv.2006.09.027>
- Brooks, M. L., Minnich, R. A., & Matchett, J. R. (2018). Southeastern Deserts Bioregion. In J. W. Van Wagtendonk, N. G. Sugihara, S. L. Stephens, A. E. Thode, K. E. Shaffer, & J. A. Fites-Kaufman (Eds.), *Fire in California's Ecosystems* (2nd ed., pp. 353–378). University of California Press; JSTOR.
<http://www.jstor.org.ezproxy.library.yorku.ca/stable/10.1525/j.ctv1wxrxh.23>
- Brooks, M. L., & Pyke, D. A. (2001). Invasive plants and fire in the deserts of North America. In K. E. M. Galley & T. P. Wilson (Eds.), *Proceedings of the Invasive Species Workshop: The Role of Fire in the Control and Spread of Invasive Species* (pp. 1–14). The First National Congress on Fire Ecology, Prevention, and Management.
- Brooks, M. L., & Zouhar, K. (2008). Plant invasions and fire regimes. In *Wildland fire in ecosystems, fire and nonnative invasive plants* (pp. 33–45). US Department of Agriculture, Forest Service.
- Brown, D. E., & Minnich, R. A. (1986). Fire and Changes in Creosote Bush Scrub of the Western Sonoran Desert, California. *The American Midland Naturalist*, 116(2), 411–422.
<https://doi.org/10.2307/2425750>
- Busetto, L., & Ranghetti, L. (2016). MODISstp: An R package for automatic preprocessing of MODIS Land Products time series. *Computers & Geosciences*, 97, 40–48.
<https://doi.org/10.1016/j.cageo.2016.08.020>
- California Natural Diversity Database (CNDDB). (2023). *State and Federally Listed Endangered and Threatened Animals of California*. California Department of Fish and Wildlife.
<https://wildlife.ca.gov/Conservation/CESA>

- Callaghan, C. T., Rowley, J. J. L., Cornwell, W. K., Poore, A. G. B., & Major, R. E. (2019). Improving big citizen science data: Moving beyond haphazard sampling. *PLOS Biology*, 17(6), e3000357. <https://doi.org/10.1371/journal.pbio.3000357>
- Carroll, K. A., Farwell, L. S., Pidgeon, A. M., Razenkova, E., Gudex-Cross, D., Helmers, D. P., Lewińska, K. E., Elsen, P. R., & Radeloff, V. C. (2022). Mapping breeding bird species richness at management-relevant resolutions across the United States. *Ecological Applications*, 32(6), e2624. <https://doi.org/10.1002/eap.2624>
- Casady, G. M., Van Leeuwen, W. J. D., & Reed, B. C. (2013). Estimating Winter Annual Biomass in the Sonoran and Mojave Deserts with Satellite- and Ground-Based Observations. *Remote Sensing*, 5(2), Article 2. <https://doi.org/10.3390/rs5020909>
- Chao, A., Chazdon, R. L., Colwell, R. K., & Shen, T.-J. (2005). A new statistical approach for assessing similarity of species composition with incidence and abundance data. *Ecology Letters*, 8(2), 148–159. <https://doi.org/10.1111/j.1461-0248.2004.00707.x>
- Chao, A., Ma, K. H., Hsieh, T. C., & Chiu, C. H. (2016). *SpadeR (Species-richness Prediction And Diversity Estimation in R)* (0.1.1) [R Package]. <https://CRAN.R-project.org/package=SpadeR>
- Chen, I.-C., Hill, J. K., Ohlemüller, R., Roy, D. B., & Thomas, C. D. (2011). Rapid Range Shifts of Species Associated with High Levels of Climate Warming. *Science*, 333(6045), 1024–1026. <https://doi.org/10.1126/science.1206432>
- Chock, R. Y., McCullough Hennessy, S., Wang, T. B., Gray, E., & Shier, D. M. (2020). A multi-model approach to guide habitat conservation and restoration for the endangered San Bernardino kangaroo rat. *Global Ecology and Conservation*, 21, e00881. <https://doi.org/10.1016/j.gecco.2019.e00881>
- Clapp, J. G., & Beck, J. L. (2016). Short-Term Impacts of Fire-Mediated Habitat Alterations on an Isolated Bighorn Sheep Population. *Fire Ecology*, 12(3), 80–98. <https://doi.org/10.4996/fireecology.1203080>

- Conway, C. J., Nadeau, C. P., & Piest, L. (2010). Fire helps restore natural disturbance regime to benefit rare and endangered marsh birds endemic to the Colorado River. *Ecological Applications*, 20(7), 2024–2035. <https://doi.org/10.1890/09-1624.1>
- Cook, E. R., Seager, R., Heim Jr, R. R., Vose, R. S., Herweijer, C., & Woodhouse, C. (2010). Megadroughts in North America: Placing IPCC projections of hydroclimatic change in a long-term palaeoclimate context. *Journal of Quaternary Science*, 25(1), 48–61. <https://doi.org/10.1002/jqs.1303>
- Cook, E. R., Woodhouse, C. A., Eakin, C. M., Meko, D. M., & Stahle, D. W. (2004). Long-Term Aridity Changes in the Western United States. *Science*, 306(5698), 1015–1018. <https://doi.org/10.1126/science.1102586>
- Crall, A. W., Jarnevich, C. S., Young, N. E., Panke, B. J., Renz, M., & Stohlgren, T. J. (2015). Citizen science contributes to our knowledge of invasive plant species distributions. *Biological Invasions*, 17(8), 2415–2427. <https://doi.org/10.1007/s10530-015-0885-4>
- Crowdsourcing and Citizen Science Act, 15 U.S.C. § § 3724 (2017). <https://www.govinfo.gov/content/pkg/USCODE-2016-title15/pdf/USCODE-2016-title15-chap63-sec3724.pdf>
- Cypher, B. L., Phillips, S. E., & Kelly, P. A. (2013). Quantity and distribution of suitable habitat for endangered San Joaquin kit foxes: Conservation implications. *Canid Biology & Conservation*, 16(7), 25–31.
- D’Antonio, C. M., & Vitousek, P. M. (1992). Biological Invasions by Exotic Grasses, the Grass/Fire Cycle, and Global Change. *Annual Review of Ecology and Systematics*, 23, 63–87.
- DeFalco, L. A., Esque, T. C., Scoles-Sciulla, S. J., & Rodgers, J. (2010). Desert Wildfire and Severe Drought Diminish Survivorship of the Long-Lived Joshua Tree (*Yucca brevifolia*; Agavaceae). *American Journal of Botany*, 97(2), 243–250.

- Dennison, P. E., Brewer, S. C., Arnold, J. D., & Moritz, M. A. (2014). Large wildfire trends in the western United States, 1984–2011. *Geophysical Research Letters*, 41(8), 2928–2933.
<https://doi.org/10.1002/2014GL059576>
- Dickinson, J. L., Shirk, J., Bonter, D., Bonney, R., Crain, R. L., Martin, J., Phillips, T., & Purcell, K. (2012). The current state of citizen science as a tool for ecological research and public engagement. *Frontiers in Ecology and the Environment*, 10(6), 291–297.
<https://doi.org/10.1890/110236>
- Dickson, B. G., & Beier, P. (2002). Home-Range and Habitat Selection by Adult Cougars in Southern California. *The Journal of Wildlife Management*, 66(4), 1235–1245.
<https://doi.org/10.2307/3802956>
- Diffenbaugh, N. S., Swain, D. L., & Touma, D. (2015). Anthropogenic warming has increased drought risk in California. *Proceedings of the National Academy of Sciences*, 112(13), 3931–3936. <https://doi.org/10.1073/pnas.1422385112>
- Diffendorfer, J., Fleming, G. M., Tremor, S., Spencer, W., & Beyers, J. L. (2012). The role of fire severity, distance from fire perimeter and vegetation on post-fire recovery of small-mammal communities in chaparral. *International Journal of Wildland Fire*, 21(4), 436.
<https://doi.org/10.1071/WF10060>
- Doherty, T. S., Geary, W. L., Jolly, C. J., Macdonald, K. J., Miritis, V., Watchorn, D. J., Cherry, M. J., Conner, L. M., González, T. M., Legge, S. M., Ritchie, E. G., Stawski, C., & Dickman, C. R. (2022). Fire as a driver and mediator of predator–prey interactions. *Biological Reviews*, 97(4), 1539–1558. <https://doi.org/10.1111/brv.12853>
- Dorph, A., Swan, M., Rochelmeyer, E., & Di Stefano, J. (2020). Complex habitat drives mammal communities in a flammable landscape. *Forest Ecology and Management*, 462, 117979.
<https://doi.org/10.1016/j.foreco.2020.117979>

- Drake, K. K., Esque, T. C., Nussear, K. E., Defalco, L. A., Scoles-Sciulla, S. J., Modlin, A. T., & Medica, P. A. (2015). Desert tortoise use of burned habitat in the Eastern Mojave desert. *The Journal of Wildlife Management*, 79(4), 618–629. <https://doi.org/10.1002/jwmg.874>
- Escuin, S., Navarro, R., & Fernández, P. (2008). Fire severity assessment by using NBR (Normalized Burn Ratio) and NDVI (Normalized Difference Vegetation Index) derived from LANDSAT TM/ETM images. *International Journal of Remote Sensing*, 29(4), 1053–1073. <https://doi.org/10.1080/01431160701281072>
- Esque, T. C., Schwalbe, C. R., DeFalco, L. A., Duncan, R. B., & Hughes, T. J. (2003). Effects of Desert Wildfires on Desert Tortoise (*Gopherus agassizii*) and Other Small Vertebrates. *The Southwestern Naturalist*, 48(1), 103–111.
- Esque, T. C., Webb, R. H., Wallace, C. S. A., Riper, C. van, McCreedy, C., & Smythe, L. (2013). Desert Fires Fueled by Native Annual Forbs: Effects of Fire on Communities of Plants and Birds in the Lower Sonoran Desert of Arizona. *The Southwestern Naturalist*, 58(2), 223–233. <https://doi.org/10.1894/0038-4909-58.2.223>
- Esri Inc. (2022). *ArcGIS Pro* (v2.9.3) [Computer software]. Esri Inc. <https://www.esri.com/en-us/arcgis/products/arcgis-pro/overview>
- Farwell, L. S., Gudex-Cross, D., Anise, I. E., Bosch, M. J., Olah, A. M., Radeloff, V. C., Razenkova, E., Rogova, N., Silveira, E. M. O., Smith, M. M., & Pidgeon, A. M. (2021). Satellite image texture captures vegetation heterogeneity and explains patterns of bird richness. *Remote Sensing of Environment*, 253, 112175. <https://doi.org/10.1016/j.rse.2020.112175>
- Filazzola, A., & Lortie, C. J. (2014). A systematic review and conceptual framework for the mechanistic pathways of nurse plants. *Global Ecology and Biogeography*, 23(12), 1335–1345. <https://doi.org/10.1111/geb.12202>

- Fithian, W., Elith, J., Hastie, T., & Keith, D. A. (2015). Bias correction in species distribution models: Pooling survey and collection data for multiple species. *Methods in Ecology and Evolution*, 6(4), 424–438. <https://doi.org/10.1111/2041-210X.12242>
- Foster, C. N., Barton, P. S., Sato, C. F., MacGregor, C. I., & Lindenmayer, D. B. (2015). Synergistic interactions between fire and browsing drive plant diversity in a forest understorey. *Journal of Vegetation Science*, 26(6), 1112–1123. <https://doi.org/10.1111/jvs.12311>
- Fusco, E. J., Finn, J. T., Balch, J. K., Nagy, R. C., & Bradley, B. A. (2019). Invasive grasses increase fire occurrence and frequency across US ecoregions. *Proceedings of the National Academy of Sciences*, 116(47), 23594–23599. <https://doi.org/10.1073/pnas.1908253116>
- GBIF.org. (2022). *GBIF Occurrence Download*. The Global Biodiversity Information Facility. <https://doi.org/10.15468/DL.TQFWSC>
- Geary, W. L., Doherty, T. S., Nimmo, D. G., Tulloch, A. I. T., & Ritchie, E. G. (2020). Predator responses to fire: A global systematic review and meta-analysis. *Journal of Animal Ecology*, 89(4), 955–971. <https://doi.org/10.1111/1365-2656.13153>
- Germano, D. J. (2019). Activity and thermal biology of Blunt-nosed Leopard Lizards (*Gambelia sila*) in the San Joaquin Desert of California. *Western North American Naturalist*, 79(3), 428–440. <https://doi.org/10.3398/064.079.0311>
- Germano, D. J., & Rathbun, G. B. (2016). Home Range and Habitat Use by Blunt-nosed Leopard Lizards in the Southern San Joaquin Desert of California. *Journal of Herpetology*, 50(3), 429–434. <https://doi.org/10.1670/15-006>
- Germano, D. J., Rathbun, G. B., & Saslaw, L. R. (2001). Managing Exotic Grasses and Conserving Declining Species. *Wildlife Society Bulletin (1973-2006)*, 29(2), 551–559.

- Germano, D. J., Rathbun, G. B., & Saslaw, L. R. (2012). Effects of grazing and invasive grasses on desert vertebrates in California. *The Journal of Wildlife Management*, 76(4), 670–682.
<https://doi.org/10.1002/jwmg.316>
- Germano, D. J., Rathbun, G. B., Saslaw, L. R., & Cypher, B. L. (2021). Home range and movements of San Joaquin antelope squirrels in the San Joaquin Desert of California. *California Fish and Wildlife Journal, CESA Special Issue*, 317–328.
<https://doi.org/10.51492/cfwj.cesasi.19>
- Germano, D. J., Rathbun, G. B., Saslaw, L. R., Cypher, B. L., Cypher, E. A., & Vredenburg, L. M. (2011). The San Joaquin Desert of California: Ecologically Misunderstood and Overlooked. *Natural Areas Journal*, 31(2), 138–147.
<https://doi.org/10.3375/043.031.0206>
- Ghazian, N., Zuliani, M., & Lortie, C. J. (2020). Micro-Climatic Amelioration in a California Desert: Artificial Shelter Versus Shrub Canopy. *Journal of Ecological Engineering*, 21(8), 216–228. <https://doi.org/10.12911/22998993/126875>
- Gray, M. E., Dickson, B. G., Zachmann, L. J., Gray, M. E., Dickson, B. G., & Zachmann, L. J. (2014). Modelling and mapping dynamic variability in large fire probability in the lower Sonoran Desert of south-western Arizona. *International Journal of Wildland Fire*, 23(8), 1108–1118. <https://doi.org/10.1071/WF13115>
- Green, D. S., Roloff, G. J., Heath, B. R., & Holekamp, K. E. (2015). Temporal dynamics of the responses by African mammals to prescribed fire. *The Journal of Wildlife Management*, 79(2), 235–242. <https://doi.org/10.1002/jwmg.827>
- Hereford, R., Webb, R. H., & Longpré, C. I. (2006). Precipitation history and ecosystem response to multidecadal precipitation variability in the Mojave Desert region, 1893–2001. *Journal of Arid Environments*, 67, 13–34.
<https://doi.org/10.1016/j.jaridenv.2006.09.019>

- Hijmans, R. J. (2021). *Raster: Geographic Data Analysis and Modeling. R package version 3.4-10*. [Computer software]. <https://rspatial.org/raster>
- Holl, S. A., & Bleich, V. C. (2012). *Responses of bighorn sheep and mule deer to fire and rain in the San Gabriel Mountains, California* (V. C. Bleich & D. Whittaker, Eds.; pp. 138–156). Wild Sheep Foundation; <https://www.frames.gov/catalog/17967>.
<http://media.nwsgc.org/proceedings/MWSGC-2010/Holl%20&%20Bleich%202012,%20138-156.pdf>
- Horn, K. J., McMillan, B. R., & St. Clair, S. B. (2012). Expansive fire in Mojave Desert shrubland reduces abundance and species diversity of small mammals. *Journal of Arid Environments*, 77, 54–58. <https://doi.org/10.1016/j.jaridenv.2011.10.003>
- Horn, K. J., & St. Clair, S. (2017). Wildfire and exotic grass invasion alter plant productivity in response to climate variability in the Mojave Desert. *Landscape Ecology*, 32(3), 635–646. <https://doi.org/10.1007/s10980-016-0466-7>
- Horn, K. J., Wilkinson, J., White, S., & St. Clair, S. B. (2015). Desert wildfire impacts on plant community function. *Plant Ecology*, 216(12), 1623–1634.
- Hughes, A. C., Orr, M. C., Ma, K., Costello, M. J., Waller, J., Provoost, P., Yang, Q., Zhu, C., & Qiao, H. (2021). Sampling biases shape our view of the natural world. *Ecography*, 44(9), 1259–1269. <https://doi.org/10.1111/ecog.05926>
- Humphrey, R. R. (1974). Fire in the Deserts and Desert Grassland of North America. In *Fire and Ecosystems* (pp. 365–400). Elsevier. <https://doi.org/10.1016/B978-0-12-424255-5.50016-X>
- Hunter, M. E., Omi, P. N., Martinson, E. J., & Chong, G. W. (2006). Establishment of non-native plant species after wildfires: Effects of fuel treatments, abiotic and biotic factors, and post-fire grass seeding treatments. *International Journal of Wildland Fire*, 15(2), 271. <https://doi.org/10.1071/WF05074>

- Inman, R., Esque, T., Nussear, K., Leitner, P., Matocq, M., Weisberg, P., Dilts, T., & Vandergast, A. (2013). Is there room for all of us? Renewable energy and *Xerospermophilus mohavensis*. *Endangered Species Research*, 20(1), 1–18. <https://doi.org/10.3354/esr00487>
- Isaac, N. J. B., Strien, A. J., August, T. A., Zeeuw, M. P., & Roy, D. B. (2014). Statistics for citizen science: Extracting signals of change from noisy ecological data. *Methods in Ecology and Evolution*, 5(10), 1052–1060. <https://doi.org/10.1111/2041-210X.12254>
- Ivanova, N. V., & Shashkov, M. P. (2021). *The Possibilities of GBIF Data Use in Ecological Research*. 52(1), 8.
- Ivey, K. N., Cornwall, M., Crowell, H., Ghazian, N., Nix, E., Owen, M., Zuliani, M., Lortie, C. J., Westphal, M., & Taylor, E. (2020). Thermal ecology of the federally endangered blunt-nosed leopard lizard (*Gambelia sila*). *Conservation Physiology*, 8(1), coaa014. <https://doi.org/10.1093/conphys/coaa014>
- João, T., João, G., Bruno, M., & João, H. (2018). Indicator-based assessment of post-fire recovery dynamics using satellite NDVI time-series. *Ecological Indicators*, 89, 199–212. <https://doi.org/10.1016/j.ecolind.2018.02.008>
- Johnston, A., Matechou, E., & Dennis, E. B. (2022). Outstanding challenges and future directions for biodiversity monitoring using citizen science data. *Methods in Ecology and Evolution*, 1–14. <https://doi.org/10.1111/2041-210X.13834>
- Kelly, P. A., Phillips, S. E., & Williams, D. F. (2006). Documenting Ecological Change in Time and Space: The San Joaquin Valley of California. In E. Lacey (Ed.), *Mammalian Diversification: From Chromosomes to Phylogeography* (p. 0). University of California Press. <https://doi.org/10.1525/california/9780520098534.003.0002>
- Kerr, G. D., & Bull, C. M. (2004). Microhabitat Use by the Scincid Lizard *Tiliqua rugosa*: Exploiting Natural Temperature Gradients beneath Plant Canopies. *Journal of Herpetology*, 38(4), 536–545. <https://doi.org/10.1670/82-04A>

- Kirchhoff, C., Callaghan, C. T., Keith, D. A., Indiarto, D., Taseski, G., Ooi, M. K. J., Le Breton, T. D., Mesaglio, T., Kingsford, R. T., & Cornwell, W. K. (2021). Rapidly mapping fire effects on biodiversity at a large-scale using citizen science. *Science of The Total Environment*, 755, 142348. <https://doi.org/10.1016/j.scitotenv.2020.142348>
- Klinger, R., Underwood, E. C., McKinley, R., & Brooks, M. L. (2021). Contrasting Geographic Patterns of Ignition Probability and Burn Severity in the Mojave Desert. *Frontiers in Ecology and Evolution*, 9. <https://doi.org/10.3389/fevo.2021.593167>
- Kotler, B. P. (1984). Risk of Predation and the Structure of Desert Rodent Communities. *Ecology*, 65(3), 689–701. <https://doi.org/10.2307/1938041>
- Kozma, J. M., & Mathews, N. E. (1997). Breeding Bird Communities and Nest Plant Selection in Chihuahuan Desert Habitats in South-Central New Mexico. *The Wilson Bulletin*, 109(3), 424–436.
- Kumari, N., Saco, P., Rodriguez, J., Johnstone, S., Srivastava, A., Chun, K., & Yetemen, O. (2020). The Grass Is Not Always Greener on the Other Side: Seasonal Reversal of Vegetation Greenness in Aspect-Driven Semiarid Ecosystems. *Geophysical Research Letters*, 47. <https://doi.org/10.1029/2020GL088918>
- Lawson, B., Petrovan, S. O., & Cunningham, A. A. (2015). Citizen Science and Wildlife Disease Surveillance. *EcoHealth*, 12(4), 693–702. <https://doi.org/10.1007/s10393-015-1054-z>
- Lenoir, J., & Svenning, J.-C. (2015). Climate-related range shifts – a global multidimensional synthesis and new research directions. *Ecography*, 38(1), 15–28. <https://doi.org/10.1111/ecog.00967>
- Lenth, R. V., Bolker, B., Buerkner, P., Giné-Vázquez, I., Herve, M., Jung, M., Love, J., Miguez, F., Riebl, H., & Singmann, H. (2023). *emmeans: Estimated Marginal Means, aka Least-Squares Means* (1.8.6) [Computer software]. <https://cran.r-project.org/web/packages/emmeans/index.html>

- Lloyd, T. J., Fuller, R. A., Oliver, J. L., Tulloch, A. I., Barnes, M., & Steven, R. (2020). Estimating the spatial coverage of citizen science for monitoring threatened species. *Global Ecology and Conservation*, 23, e01048. <https://doi.org/10.1016/j.gecco.2020.e01048>
- Loarie, S. (2020, May 15). *We've reached 1,000,000 observers!* [Www.inaturalist.org]. INaturalist; iNaturalist. <https://www.inaturalist.org/blog/35758-we-ve-reached-1-000-000-observers>
- Lortie, C. J., Filazzola, A., Kelsey, R., Hart, A. K., & Butterfield, H. S. (2018). Better late than never: A synthesis of strategic land retirement and restoration in California. *Ecosphere*, 9(8), e02367. <https://doi.org/10.1002/ecs2.2367>
- Lortie, C. J., Filazzola, A., & Sotomayor, D. A. (2016). Functional assessment of animal interactions with shrub-facilitation complexes: A formal synthesis and conceptual framework. *Functional Ecology*, 30(1), 41–51. <https://doi.org/10.1111/1365-2435.12530>
- Lortie, C. J., Filazzola, A., Westphal, M., & Butterfield, H. S. (2022). Foundation plant species provide resilience and microclimatic heterogeneity in drylands. *Scientific Reports*, 12(1), Article 1. <https://doi.org/10.1038/s41598-022-22579-1>
- Lüdecke, D., Ben-Shachar, M., Patil, I., Waggoner, P., & Makowski, D. (2021). performance: An R Package for Assessment, Comparison and Testing of Statistical Models. *Journal of Open Source Software*, 6(60), 3139. <https://doi.org/10.21105/joss.03139>
- Luque, G. M., Hochberg, M. E., Holyoak, M., Hossaert, M., Gaill, F., & Courchamp, F. (2013). Ecological effects of environmental change. *Ecology Letters*, 16(s1), 1–3. <https://doi.org/10.1111/ele.12050>
- Magurran, A. E., Baillie, S. R., Buckland, S. T., Dick, J. McP., Elston, D. A., Scott, E. M., Smith, R. I., Somerfield, P. J., & Watt, A. D. (2010). Long-term datasets in biodiversity research and monitoring: Assessing change in ecological communities through time. *Trends in Ecology & Evolution*, 25(10), 574–582. <https://doi.org/10.1016/j.tree.2010.06.016>

- Mann, M. E., & Gleick, P. H. (2015). Climate change and California drought in the 21st century. *Proceedings of the National Academy of Sciences*, 112(13), 3858–3859.
<https://doi.org/10.1073/pnas.1503667112>
- Martin, A. E., & Fahrig, L. (2018). Habitat specialist birds disperse farther and are more migratory than habitat generalist birds. *Ecology*, 99(9), 2058–2066.
<https://doi.org/10.1002/ecy.2428>
- McAuliffe, J. R. (2020). Pre-Eurosettlement Wildfires in Mojave National Preserve. *Mojave National Preserve Science Newsletter*, 1–8.
- McDonald, T. L., Erickson, W. P., & McDonald, L. L. (2000). Analysis of Count Data from Before-after Control-Impact Studies. *Journal of Agricultural, Biological, and Environmental Statistics*, 5(3), 262–279. <https://doi.org/10.2307/1400453>
- McKinley, D. C., Miller-Rushing, A. J., Ballard, H. L., Bonney, R., Brown, H., Cook-Patton, S. C., Evans, D. M., French, R. A., Parrish, J. K., Phillips, T. B., Ryan, S. F., Shanley, L. A., Shirk, J. L., Stepenuck, K. F., Weltzin, J. F., Wiggins, A., Boyle, O. D., Briggs, R. D., Chapin, S. F., ... Soukup, M. A. (2017). Citizen science can improve conservation science, natural resource management, and environmental protection. *Biological Conservation*, 208, 15–28. <https://doi.org/10.1016/j.biocon.2016.05.015>
- McLaughlin, S. P., & Bowers, J. E. (1982). Effects of Wildfire on A Sonoran Desert Plant Community. *Ecology*, 63(1), 246–248. <https://doi.org/10.2307/1937048>
- Meddens, A. J. H., Kolden, C. A., Lutz, J. A., Smith, A. M. S., Cansler, C. A., Abatzoglou, J. T., Meigs, G. W., Downing, W. M., & Krawchuk, M. A. (2018). Fire Refugia: What Are They, and Why Do They Matter for Global Change? *BioScience*.
<https://doi.org/10.1093/biosci/biy103>
- Meehan, T. D., Michel, N. L., & Rue, H. (2019). Spatial modeling of Audubon Christmas Bird Counts reveals fine-scale patterns and drivers of relative abundance trends. *Ecosphere*, 10(4), e02707. <https://doi.org/10.1002/ecs2.2707>

- Mendelsohn, M. B., Brehme, C. S., Rochester, C. J., Stokes, D. C., Hathaway, S. A., & Fisher, R. N. (2008). Responses in Bird Communities to Wildland Fires in Southern California. *Fire Ecology*, 4(2), 63–82. <https://doi.org/10.4996/fireecology.0402063>
- Mendelsohn, M. B., Madden-Smith, M. C., & Fisher, R. N. (2005). *Post-Cedar Fire Arroyo Toad (Bufo californicus) Monitoring Surveys at Cuyamaca Rancho State Park, 2004* (Technical Report, p. 42 pp) [Final Report]. USGS.
- Michael, Y. (2021). Forecasting fire risk with machine learning and dynamic information derived from satellite vegetation index time-series. *Science of the Total Environment*.
- Minnich, R. A. (2018). California Fire Climate. In J. W. Van Wagendonk, N. G. Sugihara, S. L. Stephens, A. E. Thode, K. E. Shaffer, Kevin E., & J. A. Fites-Kaufman (Eds.), *Fire in California's Ecosystems* (2nd ed., pp. 11–26). University of California Press; JSTOR. <http://www.jstor.org.ezproxy.library.yorku.ca/stable/10.1525/j.ctv1wxrxh.7>
- Moloney, K. A., Mudrak, E. L., Fuentes-Ramirez, A., Parag, H., Schat, M., & Holzapfel, C. (2019). Increased fire risk in Mojave and Sonoran shrublands due to exotic species and extreme rainfall events. *Ecosphere*, 10(2), e02592. <https://doi.org/10.1002/ecs2.2592>
- Monasmith, T. J., Demarais, S., Root, J. J., & Britton, C. M. (2010). Short-Term Fire Effects on Small Mammal Populations and Vegetation of the Northern Chihuahuan Desert. *International Journal of Ecology*, 2010, e189271. <https://doi.org/10.1155/2010/189271>
- Myneni, R. B., Hall, F. G., Sellers, P. J., & Marshak, A. L. (1995). The interpretation of spectral vegetation indexes. *IEEE Transactions on Geoscience and Remote Sensing*, 33(2), 481–486. <https://doi.org/10.1109/TGRS.1995.8746029>
- National Science Foundation. (2023). *Proposal and Award Policies and Procedures Guide*. https://nsf-gov-resources.nsf.gov/2022-10/nsf23_1.pdf?VersionId=7yfhel.bNrekBK7F5cKu9riXFbi1YjRX

- Neate-Clegg, M. H. C., Horns, J. J., Adler, F. R., Kemahlı Aytekin, M. Ç., & Şekercioğlu, Ç. H. (2020). Monitoring the world's bird populations with community science data. *Biological Conservation*, 248, 108653. <https://doi.org/10.1016/j.biocon.2020.108653>
- Newman, G., Chandler, M., Clyde, M., McGreavy, B., Haklay, M., Ballard, H., Gray, S., Scarpino, R., Hauptfeld, R., Mellor, D., & Gallo, J. (2017). Leveraging the power of place in citizen science for effective conservation decision making. *Biological Conservation*, 208, 55–64. <https://doi.org/10.1016/j.biocon.2016.07.019>
- O'Connor, M. P., Zimmerman, L. C., Ruby, D. E., Bulova, S. J., & Spotila, J. R. (1994). Home Range Size and Movements by Desert Tortoises, *Gopherus agassizii*, in the Eastern Mojave Desert. *Herpetological Monographs*, 8, 60–71. <https://doi.org/10.2307/1467070>
- Omernik, J. M., & Griffith, G. E. (2014). Ecoregions of the Conterminous United States: Evolution of a Hierarchical Spatial Framework. *Environmental Management*, 54(6), 1249–1266. <https://doi.org/10.1007/s00267-014-0364-1>
- Park, I. W., Mann, M. L., Flint, L. E., Flint, A. L., & Moritz, M. (2021). Relationships of climate, human activity, and fire history to spatiotemporal variation in annual fire probability across California. *PLOS ONE*, 16(11), e0254723. <https://doi.org/10.1371/journal.pone.0254723>
- Pastro, L. A., Dickman, C. R., & Letnic, M. (2011). Burning for biodiversity or burning biodiversity? Prescribed burn vs. wildfire impacts on plants, lizards, and mammals. *Ecological Applications*, 21(8), 3238–3253. <https://doi.org/10.1890/10-2351.1>
- Pausas, J. G. (2019). Generalized fire response strategies in plants and animals. *Oikos*, 128(2), 147–153. <https://doi.org/10.1111/oik.05907>
- Pebesma, E. (2018). Simple Features for R: Standardized Support for Spatial Vector Data. *The R Journal*, 10(1), 439. <https://doi.org/10.32614/RJ-2018-009>
- Pecl, G. T., Araújo, M. B., Bell, J. D., Blanchard, J., Bonebrake, T. C., Chen, I.-C., Clark, T. D., Colwell, R. K., Danielsen, F., Evengård, B., Falconi, L., Ferrier, S., Frusher, S., Garcia,

- R. A., Griffis, R. B., Hobday, A. J., Janion-Scheepers, C., Jarzyna, M. A., Jennings, S., ... Williams, S. E. (2017). Biodiversity redistribution under climate change: Impacts on ecosystems and human well-being. *Science*, 355(6332), eaai9214.
<https://doi.org/10.1126/science.aai9214>
- Pettorelli, N., Vik, J. O., Mysterud, A., Gaillard, J.-M., Tucker, C. J., & Stenseth, N. Chr. (2005). Using the satellite-derived NDVI to assess ecological responses to environmental change. *Trends in Ecology & Evolution*, 20(9), 503–510.
<https://doi.org/10.1016/j.tree.2005.05.011>
- Pidgeon, A. M., Radeloff, V. C., & Mathews, N. E. (2003). Landscape-Scale Patterns of Black-Throated Sparrow (*amphispiza Bilineata*) Abundance and Nest Success. *Ecological Applications*, 13(2), 530–542. [https://doi.org/10.1890/1051-0761\(2003\)013\[0530:LSPOBT\]2.0.CO;2](https://doi.org/10.1890/1051-0761(2003)013[0530:LSPOBT]2.0.CO;2)
- Poloczanska, E. S., Brown, C. J., Sydeman, W. J., Kiessling, W., Schoeman, D. S., Moore, P. J., Brander, K., Bruno, J. F., Buckley, L. B., Burrows, M. T., Duarte, C. M., Halpern, B. S., Holding, J., Kappel, C. V., O'Connor, M. I., Pandolfi, J. M., Parmesan, C., Schwing, F., Thompson, S. A., & Richardson, A. J. (2013). Global imprint of climate change on marine life. *Nature Climate Change*, 3(10), Article 10.
<https://doi.org/10.1038/nclimate1958>
- R Core Team. (2022). *R: A language and environment for statistical computing*. (v4.2.0) [Computer software]. R Foundation for Statistical Computing. <https://www.R-project.org/>
- Renner, I. W., Elith, J., Baddeley, A., Fithian, W., Hastie, T., Phillips, S. J., Popovic, G., & Warton, D. I. (2015). Point process models for presence-only analysis. *Methods in Ecology and Evolution*, 6(4), 366–379. <https://doi.org/10.1111/2041-210X.12352>
- Ribeiro, I., Proença, V., Serra, P., Palma, J., Domingo-Marimon, C., Pons, X., & Domingos, T. (2019). *Mediterranean rural landscapes*. 13.

- Rieman, B., Gresswell, R., & Rinne, J. (2012). *Fire and Fish: A Synthesis of Observation and Experience* (General Technical Report RMRS-GTR-290; In: Luce, Charles; Morgan, Penny; Dwire, Kathleen; Isaak, Daniel; Holden, Zachary; Rieman, Bruce. Climate Change, Forests, Fire, Water, and Fish: Building Resilient Landscapes, Streams, and Managers., pp. 159–175). U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station.
- Salvatori, V., Eguny, F., Skidmore, A. K., De Leeuw, J., & Van Gils, H. a. M. (2001). The effects of fire and grazing pressure on vegetation cover and small mammal populations in the Maasai Mara National Reserve. *African Journal of Ecology*, 39(2), 200–204.
<https://doi.org/10.1046/j.1365-2028.2001.00295.x>
- Saracco, J. F., Fettig, S. M., Miguel, G. L. S., Mehlman, D. W., Thompson, B. E., & Albert, S. K. (2018). Avian demographic responses to drought and fire: A community-level perspective. *Ecological Applications*, 28(7), 1773–1781.
<https://doi.org/10.1002/eap.1751>
- Shaffer, K. E., Hedwall, S. J., & Laudenslayer, W. F. (2018). Fire and Animal Interactions. In K. E. Shaffer, J. W. van Wagtendonk, N. G. Sugihara, S. L. Stephens, A. E. Thode, & J. A. Fites-Kaufman (Eds.), *Fire in California's Ecosystems* (2nd ed., pp. 123–146). University of California Press; JSTOR.
<http://www.jstor.org.ezproxy.library.yorku.ca/stable/10.1525/j.ctv1wxrxh.14>
- Sharp Bowman, T. R., McMillan, B. R., & St. Clair, S. B. (2017). A comparison of the effects of fire on rodent abundance and diversity in the Great Basin and Mojave Deserts. *PLOS ONE*, 12(11), e0187740. <https://doi.org/10.1371/journal.pone.0187740>
- Smale, D. A., Wernberg, T., Oliver, E. C. J., Thomsen, M., Harvey, B. P., Straub, S. C., Burrows, M. T., Alexander, L. V., Benthuyssen, J. A., Donat, M. G., Feng, M., Hobday, A. J., Holbrook, N. J., Perkins-Kirkpatrick, S. E., Scannell, H. A., Sen Gupta, A., Payne, B. L., & Moore, P. J. (2019). Marine heatwaves threaten global biodiversity and the

- provision of ecosystem services. *Nature Climate Change*, 9(4), Article 4.
<https://doi.org/10.1038/s41558-019-0412-1>
- St. Clair, S. B., & Bishop, T. B. B. (2019). Loss of biotic resistance and high propagule pressure promote invasive grass-fire cycles. *Journal of Ecology*, 107(4), 1995–2005.
<https://doi.org/10.1111/1365-2745.13156>
- St. Clair, S. B., O'Connor, R., Gill, R., & McMillan, B. (2016). Biotic resistance and disturbance: Rodent consumers regulate post-fire plant invasions and increase plant community diversity. *Ecology*, 97(7), 1700–1711. <https://doi.org/10.1002/ecy.1391>
- Steers, R. J., & Allen, E. B. (2011). Fire Effects on Perennial Vegetation in the Western Colorado Desert, USA. *Fire Ecology*, 7(3), Article 3.
<https://doi.org/10.4996/fireecology.0703059>
- Steers, R. J., & Allen, E. B. (2012). Impact of Recurrent Fire on Annual Plants: A Case Study from the Western Edge of the Colorado Desert. *Madroño*, 59(1), 14–24.
<https://doi.org/10.3120/0024-9637-59.1.14>
- St-Louis, V., Pidgeon, A. M., Clayton, M. K., Locke, B. A., Bash, D., & Radeloff, V. C. (2009). Satellite image texture and a vegetation index predict avian biodiversity in the Chihuahuan Desert of New Mexico. *Ecography*, 32(3), 468–480.
<https://doi.org/10.1111/j.1600-0587.2008.05512.x>
- Stoklosa, J., Blakey, R. V., & Hui, F. K. C. (2022). An Overview of Modern Applications of Negative Binomial Modelling in Ecology and Biodiversity. *Diversity*, 14(5), Article 5.
<https://doi.org/10.3390/d14050320>
- Sugihara, N. G., Keeler-Wolf, T., & Barbour, M. G. (2018). Fire and California Vegetation. In N. G. Sugihara, J. W. Van Wagtenonk, S. L. STEPHENS, A. E. THODE, K. E. SHAFFER, & J. A. FITES-KAUFMAN (Eds.), *Fire in California's Ecosystems* (2nd ed., pp. 1–8). University of California Press; JSTOR.
<http://www.jstor.org.ezproxy.library.yorku.ca/stable/10.1525/j.ctv1wxrxh.6>

- Syphard, A. D., & Keeley, J. E. (2020). Mapping fire regime ecoregions in California. *International Journal of Wildland Fire*, 29(7), 595. <https://doi.org/10.1071/WF19136>
- Syphard, A. D., Keeley, J. E., & Abatzoglou, J. T. (2017). Trends and drivers of fire activity vary across California aridland ecosystems. *Journal of Arid Environments*, 144, 110–122. <https://doi.org/10.1016/j.jaridenv.2017.03.017>
- Team eBird. (2021, December 21). 2021 Year in Review: *EBird*, *Merlin*, *Macaulay Library*, and *Birds of the World* - eBird. Ebird.Org. <https://ebird.org/ebird/news/2021-year-in-review>
- Theobald, E. J., Ettinger, A. K., Burgess, H. K., DeBey, L. B., Schmidt, N. R., Froehlich, H. E., Wagner, C., HilleRisLambers, J., Tewksbury, J., Harsch, M. A., & Parrish, J. K. (2015). Global change and local solutions: Tapping the unrealized potential of citizen science for biodiversity research. *Biological Conservation*, 181, 236–244. <https://doi.org/10.1016/j.biocon.2014.10.021>
- Underwood, E. C., Klinger, R. C., & Brooks, M. L. (2019). Effects of invasive plants on fire regimes and postfire vegetation diversity in an arid ecosystem. *Ecology and Evolution*, 9(22), 12421–12435. <https://doi.org/10.1002/ece3.5650>
- van Mantgem, P., Schwartz, M., & Keifer, M. (2001). *Monitoring Fire Effects for Managed Burns and Wildfires: Coming to Terms with Pseudoreplication*. http://www.naturalareas.org/docs/v21_3_01_pp266_273.pdf
- van Strien, A. J., van Swaay, C. A. M., & Termaat, T. (2013). Opportunistic citizen science data of animal species produce reliable estimates of distribution trends if analysed with occupancy models. *Journal of Applied Ecology*, 50(6), 1450–1458. <https://doi.org/10.1111/1365-2664.12158>
- van Wagtenonk, J. W., & Cayan, D. R. (2008). Temporal and Spatial Distribution of Lightning Strikes in California in Relation to Large-Scale Weather Patterns. *Fire Ecology*, 4(1), Article 1. <https://doi.org/10.4996/fireecology.0401034>

- Ver Hoef, J. M., & Boveng, P. L. (2007). Quasi-Poisson vs. negative binomial regression: How should we model overdispersed count data? *Ecology*, 88(11), 2766–2772.
<https://doi.org/10.1890/07-0043.1>
- Villarreal, M. L., Norman, L. M., Buckley, S., Wallace, C. S. A., & Coe, M. A. (2016). Multi-index time series monitoring of drought and fire effects on desert grasslands. *Remote Sensing of Environment*, 183, 186–197. <https://doi.org/10.1016/j.rse.2016.05.026>
- Vitousek, P. M. (1994). Beyond Global Warming: Ecology and Global Change. *Ecology*, 75(7), 1861–1876. <https://doi.org/10.2307/1941591>
- Warrick, G. D., & Cypher, B. L. (1998). Factors Affecting the Spatial Distribution of San Joaquin Kit Foxes. *The Journal of Wildlife Management*, 62(2), 707–717.
<https://doi.org/10.2307/3802347>
- Warton, D. I., Renner, I. W., & Ramp, D. (2013). Model-Based Control of Observer Bias for the Analysis of Presence-Only Data in Ecology. *PLoS ONE*, 8(11), e79168.
<https://doi.org/10.1371/journal.pone.0079168>
- Weiss, J. L., Gutzler, D. S., Coonrod, J. E. A., & Dahm, C. N. (2004). Long-term vegetation monitoring with NDVI in a diverse semi-arid setting, central New Mexico, USA. *Journal of Arid Environments*, 58(2), 249–272. <https://doi.org/10.1016/j.jaridenv.2003.07.001>
- Whittaker, R. H. (1975). *Communities and ecosystems* (2nd ed.). Macmillan Publishing Co., Inc.
- Williams, S. E., Hobday, A. J., Falconi, L., Hero, J.-M., Holbrook, N. J., Capon, S., Bond, N. R., Ling, S. D., & Hughes, L. (2020). Research priorities for natural ecosystems in a changing global climate. *Global Change Biology*, 26(2), 410–416.
<https://doi.org/10.1111/gcb.14856>
- Wills, R. (2018). Central Valley Bioregion. In J. W. Van Wagtendonk, N. G. Sugihara, S. L. Stephens, A. E. Thode, Shaffer, Kevin E., & F.-K. Joann (Eds.), *Fire in California's Ecosystems* (2nd ed., pp. 279–298). University of California Press; JSTOR.
<http://www.jstor.org.ezproxy.library.yorku.ca/stable/10.1525/j.ctv1wxrxh.20>

Woodhouse, C. A., Meko, D. M., MacDonald, G. M., Stahle, D. W., & Cook, E. R. (2010). A 1,200-year perspective of 21st century drought in southwestern North America.

Proceedings of the National Academy of Sciences, 107(50), 21283–21288.

<https://doi.org/10.1073/pnas.0911197107>

Zuliani, M., Ghazian, N., Owen, M., Westphal, M. F., Butterfield, H. S., & Lortie, C. J. (2023).

Shrub density effects on the presence of an endangered lizard of the Carrizo Plain

National Monument, California. *Ecology and Evolution*, 13(5), e10128.

<https://doi.org/10.1002/ece3.10128>

Zuur, A. F., Ieno, E. N., Walker, N., Saveliev, A. A., & Smith, G. M. (2009). *Mixed effects models and extensions in ecology with R*. Springer New York. [https://doi.org/10.1007/978-0-387-](https://doi.org/10.1007/978-0-387-87458-6)

87458-6

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 Date (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
	COTTONWOOD D2	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

Appendix B: Supplementary material evaluating the influence of road density on mean annual ES occurrences reported between 1995-2020.

Methods

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>). The Line Density spatial analyst tool in ArcGIS Pro was used to calculate the distance to the nearest road for each ES occurrence reported (v2.9.3; Esri Inc., 2022). A linear regression fit to test the influence of road density to the mean annual ES occurrences reported between 1995 and 2020.

Results

There is a positive correlation between road density and the number of ES occurrences reported (Figure 6, $p = 0.073$).

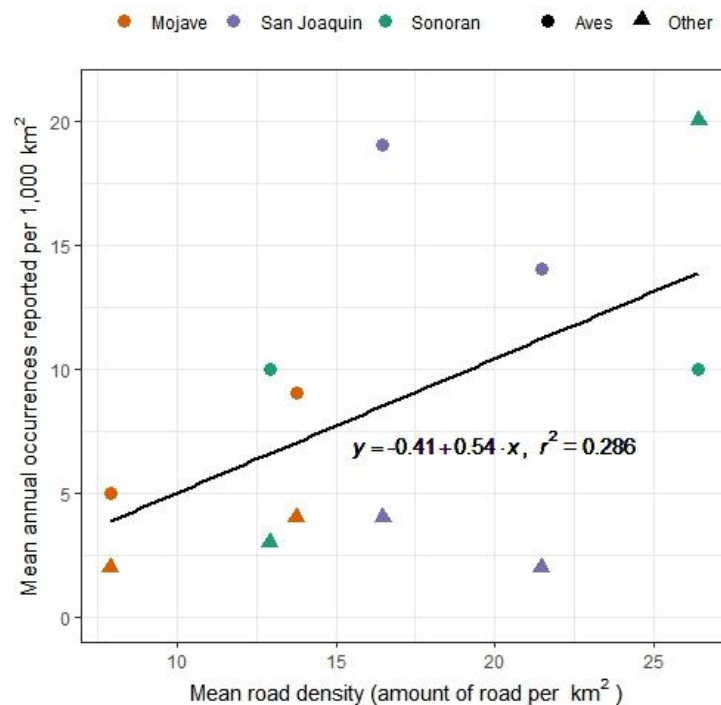


Figure 6: Relationship between mean road density and mean annual ES occurrences reported per 1,000 km² between 1995 and 2020.