USING NON-INVASIVE METHODS TO SAMPLE MAMMALIAN SPECIES IN A
POST FIRE LANDSCAPE

by

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2

3

ACKNOWLEDGMENTS

TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

ABSTRACT

Wildfires are a key process in many ecosystems worldwide. However, the intensity and severity of wildfire events has increased in recent years due to factors such as anthropogenic fire suppression and more arid conditions created by climate change. The high elevation forests of the San Bernardino Mountains in Southern California are one such ecosystem that has experienced several fire events in recent years. These forests are important habitats for a number of mammalian species. I used two non-invasive methods, camera trapping and hair tube sampling, to survey mammalian species in a recently burned area. I detected 14 species including an isolated subspecies of Humboldt's flying squirrel (*Glaucomys oregonensis*), the San Bernardino flying squirrel (*G. oregonensis californicus*). Our study showed that landscape use by large bodied and generalist species was not drastically impacted by this fire. Of the nine most commonly detected species all were detected across all burn severities but were detected more often at lower burn severities. Occupancy modeling for the forest specialist, the San Bernardino flying squirrel, showed that they can persist in burned areas if key habitat features such as an intact forest canopy and large live trees are retained. My study illustrates the ability of mammalian species to persist in fire affected areas especially when patches of forest remain intact.

INTRODUCTION

The intensity and frequency of fire events has increased worldwide in recent years. These increases can be attributed to anthropogenic factors, such as fire suppression leading to an accumulation of woody fuels (Westerling et al. 2006), and climate change creating more arid conditions (Westerling et al. 2006). Prior to the 20th century, wildfires occurred less frequently and at lower severities (Brown and Smith 2000). Milder burns created a mosaic of burn severities, compared to uniform, high intensity burns of modern fires (Bonnicksen and Stone 1982, Halofsky et al. 2011). This increase in fire severity can have dire consequences for wildlife by destroying available habitat (Fisher and Wilkinson 2005) and increasing mortality (Koprowski et al. 2006).

Mammals are a globally distributed taxon and can react to fire in various ways. Generalist species, such as California ground squirrels (*Otospermophilus beecheyi*), are found in burned areas (Converse et al. 2006) much more often than forest specialists like flying squirrels (*Glaucomys spp.,* Roberts et al. 2015, Smith 2007). Large mammals, such as ungulates, can also readily utilize burned areas (Hobbs and Spowart 1984). Where species inhabit can also affect how they react to fire, with isolated species being prone to more negative effects (Fahrig and Merriam 1985, 1994, Stacey and Taper 1992).

The San Bernardino Mountains in Southern California are home to a number of mammalian species. This mountain range has burned several times in the past few decades, with the regularity and severity of burns expected to increase with climate change (U.S. Global Change Research Program 2009, Yoon et al. 2015). In many cases the mammalian species that inhabit this range are separated from conspecific populations by considerable distances. A unique subspecies of Humboldt's flying squirrel (*Glaucomys*

oregonensis), the San Bernardino flying squirrel (*G. oregonensis californicus*), is found only in this mountain range, with the nearest population of Humboldt's flying squirrel being located 241 km away in the Sierra Nevada Mountains, California (Butler et al. 1991). This isolated, forest specialized species can be drastically impacted by wildfire (Roberts et al. 2015, Smith 2007).

Surveys of mammalian species have benefited from the use of non-invasive methods. Camera traps can be used long term and minimize human presence, which can increase detections of cryptic species (Nakashima et al. 2018, Patten and Burger 2018). Hair snares, such as hair tubes, can easily collect hair samples for microscopic, or genetic analysis, while eliminating the need to capture and handle animals (Taylor 1985, Teerink 1991, De Marinis and Asprea 2006).

In this study I aimed to assess the effect of fire on mammalian species, in an isolated mountain range. An assessment of habitat characteristics that can affect occupancy of San Bernardino flying squirrels was also conducted to help inform management decisions for this subspecies.

PRESENT STUDY

This thesis includes two manuscripts intended for publication in peer-reviewed journals. All methods, results, and conclusions of the study are included as appendices to this thesis. The first manuscript (Appendix A) titled "Response to fire by a forest specialist in isolated montane forest" is intended for submission to *Forest Ecology and Management*. This paper looks at the effects fire has on the occupancy of an isolated forest specialist, the San Bernardino flying squirrel (*Glaucomys oregonensis californicus*). The second manuscript (Appendix B) titled "A comparison of non-invasive methods for sampling species richness in a fire effected landscape" is intended for submission to *Fire Ecology*. This paper looks at the species richness across a mosaic burn using and comparing two non-invasive sampling methods, camera traps and hair tubes.

LITERATURE CITED

BONNICKSEN, T.M., AND E.C. STONE. 1982. Reconstruction of a presettlement giant

    sequoia-mixed conifer forest community using the aggregation approach. Ecology

    63:1134-1148.

BROWN, J. K., AND J. K. SMITH, EDS. 2000. Wildland fire in ecosystems: effects of fire on

    flora. United States Department of Agriculture Forest Service, Rocky Mountain

    Research Station, Ogden, Utah, RMRS-GTR-42-vol. 2.

BUTLER, R., C. SCHIFFER, AND A. MANN. 1991. Final report--San Bernardino flying

    squirrel (Conducted under a cooperative agreement with Bear Mtn LTD), United

    States Department of Agriculture Forest Service, San Bernardino Ranger District

    Final Report.

CONVERSE, S.J., W.M. BLOCK, AND G.C. WHITE. 2006. Small mammal population and

    habitat responses to forest thinning and prescribed fire. Forest Ecology and

    Management 228:263-273.

DE MARINIS, A. M., AND A. ASPREA. 2006. Hair identification key of wild and domestic

    ungulates from southern Europe. Wildlife Biology 12:305–320.

FISHER, J. T., AND L. WILKINSON. 2005. The response of mammals to forest fire and

    timber harvest in the North American boreal forest. Mammal Review 35:51–81.

FAHRIG, L., AND G. MERRIAM. 1985. Habitat patch connectivity and population survival.

    Ecology 66:1762-1768.

FAHRIG, L., AND G. MERRIAM. 1994. Conservation of fragmented populations.

    Conservation Biology 8:50-59.

HALOFSKY, J.E., D.C. DONATO, D.E. HIBBS, J.L. CAMPBELL, M.D. CANNON, J.B.
FONTAINE, J.R. THOMPSON, R.G. ANTHONY, B.T. BOREMANN, L.J. KAYES, B.E.
LAW, D.L. PETERSON, AND T.A. SPIES. 2011. Mixed-severity fire regimes: lessons
and hypotheses from the Klamath-Siskiyou Ecoregion. Ecosphere 2:40.

HOBBS, N.T. AND R.A. SPOWART. 1984. Effects of prescribed fire on nutrition of
mountain sheep and mule deer during winter and spring. The Journal of Wildlife
Management 48:551-560.

KOPROWSKI, J. L., K. M. LEONARD, C. A. ZUGMEYER, AND J. L. JOLLEY. 2006. Direct
effects of fire on endangered Mount Graham red squirrels. Southwestern
Naturalist 51:59–63.

NAKASHIMA, Y., K. FUKASWA, AND H. SAMEHIMA. 2018 Estimating animal density
without individual recognition using information derivable exclusively from
camera traps. Journal of Applied Ecology 55:735-744.

PATTEN, M. A., AND J. C. BURGER. 2018. Reserves as double-edged sword: Avoidance
behavior in an urban-adjacent wildland. Biological Conservation 218:233-239.

ROBERTS, S.L., D.A. KELT, J.W. VAN WAGTENDONK, A.K. MILES, AND M.D. MEYER.
2015. Effects of fire on small mammal communities in frequent-fire forests in
California. Journal of Mammalogy 96:107-119.

SMITH, W. P. 2007. Ecology of *Glaucomys sabrinus*: habitat, demography, and
community relations. Journal of Mammalogy 88:862-881.

STACEY, P. B., AND M. TAPER. 1992. Environmental variation and the persistence of
small populations. Ecological Applications 2:18-29.

TAYLOR, R. J. 1985. Identification of the hair of Tasmanian mammals. Papers and
    Proceedings of the Royal Society of Tasmania 119:69–82.

TEERINK, B.J. 1991. Hair of West European Mammals: Atlas and Identification Key.
    Cambridge University Press, Cambridge, UK.

U.S. GLOBAL CHANGE RESEARCH PROGRAM. 2009. Global climate change impacts in the
    United States. U.S. Global Change Research Program. Thomas R. Karl, Jerry M.
    Melillo, and Thomas C. Peterson, (eds.). Cambridge University Press, 2009.

WESTERLING, A. L., H. G. HIDALGO, D. R. CAYAN, AND T. W. SWETNAM. 2006. Warming
    and earlier spring increase western US forest wildfire activity. Science 313:940–
    943.

YOON, J, S. WANG, R. R. GILLIES, L. HIPPS, B. KRAVIT, AND P. J. RASCH. 2015. Extreme
    fire season in California: A glimpse into the future. Bulletin of the American
    Meteorological Society 96:S1-5.

APPENDIX A: RESPONSE TO FIRE BY A FOREST SPECIALIST IN ISOLATED MONTANE FOREST

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This article will be submitted to *Forest Ecology and Management* for publication.

RESPONSE TO FIRE BY A FOREST SPECIALIST IN ISOLATED MONTANE FOREST

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ABSTRACT--Fire events have increased in scale and severity due to hotter, drier conditions brought on by climate change and fire suppression. Extreme fire events can be detrimental to forest specialists, especially populations at the edge of the species range, where conditions can vary from those within the core of the range. The San Bernardino flying squirrel (*Glaucomys oregonensis californicus*) is the southernmost subspecies of Humboldt's flying squirrel, and occurs only in the San Bernardino Mountains of southern California after apparent extirpation from the nearby San Jacinto Mountains. We used non-invasive methods to assess how fire affects flying squirrel occupancy. We surveyed for flying squirrels in burned and unburned areas of the San Bernardino Mountains, and measured habitat features to construct occupancy models. Our top models indicated that flying squirrels occupy lower burn severities but could still occur at higher burn severities if canopy cover remained intact, and tree mortality remained low, especially on steeper slopes. Our study illustrates the ability of San Bernardino flying squirrels, and the potential for other flying squirrel species and subspecies, to survive major fire events.

Such results are especially important for species management as fire frequency and severity is expected to increase.

1. INTRODUCTION--Anthropogenic influences such as urban development and climate change have impacted several forest-dwelling species worldwide. Climate change has a wide range of effects, including decreases in precipitation (Kelly and Goulden 2008, Trenberth et al. 2007, Hayhoe et al. 2004), and increases in temperature (Bonfils et al. 2008a, 2008b, Cayan et al. 2008), which can cause shifts in species ranges (Parmesan and Yohe 2003, Parmesan 2006). Populations at the periphery of the species range can provide insight into how populations under different conditions may react as climatic conditions continue to change (Hampe and Petit 2005, Gigliotti et al. 2017). These same peripheral populations, especially those that are isolated, are often at the greatest risk of extirpation from environmental changes brought on by climate change (Aitken et al. 2008, Pearson et al. 2009). When a changing environment is paired with a disturbance event, such as a wildfire, it can be catastrophic for vulnerable populations. Historically, fires across the United States burned more frequently and at lower intensities and played a key role in maintaining forest ecosystems (Brown and Smith 2000). Over the past century wildfire suppression has become a common forestry practice, and has led to a number of negative effects for forest health and wildlife (Czech et al. 2000, Bentz and Woodard 1988, Wilsen et al. 1995). Modern wildfires in the United States tend to burn with a greater severity and scale due to the hotter, drier conditions caused by climate change (Westerling et al. 2006), fire suppression, and accrued fuels (Westerling et al. 2006) leading to increased animal mortality (Koprowski et al. 2006) and degradation of

habitat (Fisher and Wilkinson 2005). Species that depend on mature forests, such as the Humboldt's flying squirrel (*Glaucomys oregonensis*), are often most negatively affected by fire and climate change due to the alteration and long recovery time of forest ecosystems upon which squirrels depend (Carey 1995, Lehmkuhl et al. 2006, Smith 2007, Waters and Zabel 1995).

The San Bernardino flying squirrel (*G. oregonensis californicus*) is the southernmost subspecies of Humboldt's flying squirrel. The historical range of the subspecies included high elevation forests of the San Bernardino and San Jacinto mountains in Southern California. However, the San Bernardino flying squirrel is currently only found in the San Bernardino Mountains and is believed to have been extirpated from the San Jacinto Mountains, as studies of owl pellets did not detect squirrels (Stephenson and Calcorone 1999) and the last confirmed sighting occurring in the 1990s (Federal Register 2016). The nearest conspecific population of Humboldt's flying squirrels is 241 km away in the Sierra Nevada Mountains, California (Butler et al. 1991). The isolation of this subspecies paired with the different habitat conditions in the San Bernardino Mountains makes the San Bernardino flying squirrel vulnerable to population decline from the effects of climate change and forest fires. Climate change is likely to reduce the available habitat in the San Bernardino Mountains (U.S. Global Change Research Program 2009) and produce hotter, drier conditions that can negatively affect truffle growth the squirrels rely on for food (Smith 2007), and increase the risk of forest fires. The San Bernardino Mountains already experience a number of fire events each year, and as conditions become more arid, fire frequency and intensity are expected to increase (Yoon et al. 2015).

The 3Rs (replacement, reduction, refinement) for the ethical use of animals in research, testing, and education are as applicable today as when they were first proposed (Russel and Burch 1959). Replacement focuses on methods that allow for the comparable results to be obtained without the use of animals. Reduction refers to getting similar results using fewer animals, or more data from the same number of animals. Refinement looks to use methods that minimize animal stress, pain, suffering, and improve animal wellbeing. While the 3Rs were initially applied to laboratory studies of animals their use has also been applied to wildlife research, especially refinement (Brivio et al. 2015). The methods used to study wildlife such as live trapping, or transect surveys aim to reduce any adverse effects to their subjects from handling, sedation, or encounters with humans (Mulcay 2003, Cattet et al. 2008). With recent advances in technology new non-invasive methods for sampling wildlife have come into vogue that further reduce any potential risks. The field of genetics has especially embraced the use of non-invasive sampling. Traditionally to obtain samples for genetic analysis involve the capture and immobilization of the target animal. However, the use of hair snares (Foran et al. 1997, Mondol et al. 2009), collection of scat (Lukas and Burnham 2005, Harris et al. 2010), and environmental DNA (eDNA, Bohmann et al. 2014, Thomsen and Willerslev 2015) all allow for the collection of samples without the need to for the target species to even be seen by the researcher. Even traditional wildlife surveys benefit from technologies such as camera traps (Nakashima et al. 2018, Patten and Burger 2018) and hair snares (Taylor 1985, Teerink 1991, De Marinis and Asprea 2006) allowing researchers to identify the species in an area without being present and over much longer periods.

Our objectives were to identify key habitat features and investigate the effects of fire on occupancy of the San Bernardino flying squirrel. Our results will increase the knowledge of how peripheral populations can respond to large scale disturbance events in the environment.

2. MATERIALS AND METHODS

2.1. *Study Area*--Our study area was located in the San Bernardino Mountains in Southern California. The San Bernardino Mountains are a forested sky island that reach an elevation of about 3,500 m. Vegetation at the lowest elevations consists of chaparral and evergreen oak woodlands, which transition into mixed conifer-oak forests at higher elevations. This mountain range receives highly seasonal precipitation, mostly November-March, more often than the surrounding valleys that allow these high elevation forests to grow. Even with this increased precipitation, these montane forests are susceptible to wildfire events.

We established two study areas, one affected by a recent fire event and the second in a nearby unburned location. The fire impacted site, referred to as the burned study area, was located just west of Lake Arrowhead and was the site of 2007 Grass Valley fire that burned 366 ha, much of which was San Bernardino flying squirrel habitat. Burn severities and tree mortality data, which were collected one-year post-fire, were obtained from the U.S. Forest Service San Bernardino Ranger District. The Grass Valley fire fragmented the landscape with forest patches ranging from unburned to high burn severity. Following the fire event, USDA burn severity classifications were calculated across the study area (Parson et al. 2010). These categories range from 1-4 and are:

unburned-very low (1), low (2), moderate (3), and severe (4). Category 1 is categorized by almost all surface organics (soil, woody debris, surface roots, etc.) remaining intact with little to no charring, and with vegetation remaining green. Category 2 consists of the majority of surface organics remaining recognizable with light black or brown charring, and vegetation remaining green. In Category 3 up to 80% of surface organics are consumed leaving black charring and gray ash, and vegetation becoming brown from scorching. Category 4 is the most severe with almost all surface organics destroyed with heavy charring and more ash, with almost no vegetation remaining. We also categorized tree mortality, or the percent of vegetation killed by fire, across the study area into categories from 1-7 (1=0%, 2=1-10%, 3=10-25%, 4=25-50%, 5=50-75%, 6=75-90%, 7=>90%). The second site was established nearby in an unburned area (classified as burn severity and tree mortality 0) south of California State Route 173 and east of Grass Valley Road.

We used ArcMap (2015, v10.3.1, ESRI, Redlands, California) to create stratified random sampling sites in each study area. Within the burned area, we created 115 random sampling sites that were stratified across different burn severities (30 unburned-very low, 35 low burn, 35 moderate burn, 15 high burn). Within the unburned study area, we created 30 random sampling sites. Surveys for flying squirrels across both study areas occurred four times from June-October 2014 and five times from May-October 2015.

2.2. *Hair tube Sampling*--We used hair sampling tubes at all established random sampling sites in both study areas that acted as hair snares to assess the occupancy of San Bernardino flying squirrels (Suckling 1978, Scotts and Craig 1988, Lindenmayer 1999).

We constructed hair tubes from 30 cm long x 10 cm diameter piece of PVC pipe. We attached duct tape (3M Company, Maplewood, Minnesota) to wooden blocks located at the ends of the tubes to snag hair samples from any animal of appropriate size that entered. We baited hair tubes with whole shell peanuts and peanut butter placed in a hardware cloth basket at the center of each tube. We placed two tubes at each site, one at a height 1-5 m in a tree closest to the randomly selected site on a branch parallel to the ground (depending on tree height and branch availability) and the second on the ground near the base of the tree.

A survey was the time between visits to a site to remove tape. We collected hair samples a mean of 30.8 ($\pm$8.3) days over a total mean of 123.0 ($\pm$9.0) days in 2014 and 157.8 ($\pm$13.4) days in 2015 per site across all survey periods. Upon removal, we placed pieces of tape containing hair in labeled plastic bags with the interior coated in talcum powder to minimize adhesion to the bag. We discarded any tape that did not contain hair and recorded as no detection. After we collected hair samples, we replaced tape and bait in each hair tube.

We placed tape containing hair samples in a xylene (Crown, Packaging Service Co., Inc., Pearland, Texas) bath under a fume hood for approximately five minutes to dissolve the tape's adhesive and removed the hair. We mounted clean and dried hair samples on microscope slides with mineral oil. We used a compound light microscope (Nikon Eclipse E200, Nikon Corporation, Minato, Tokyo, Japan) at 40x magnification to view the mounted hair samples. We examined medulla and cortex characteristics of each hair (Teerink 1991) and compared to reference hair samples obtained from the University of Arizona Mammal Collection to identify species of origin, with a focus on identifying

samples from the San Bernardino flying squirrel. In some cases, species could not be determined from medulla and cortex characteristics alone, so we examined scale patterns on the outer cuticle of hair. We obtained scale patterns by coating microscope slides in a thin layer of 10-20% gelatin solution and allowed them to dry. We moistened hair samples, pressed them into the gelatin on the coated slides, and removed hairs when dry to reveal an impression of the scale pattern, which we compared to reference samples (Teerink 1991).

     2.3. *Camera Traps*--We randomly selected a subset of hair tube sites at which to install remote sensor camera traps (Bushnell 8 MP Trophy camera, model no. 119436C, Bushnell Corporation, Overland Park, Kansas) and stratified these across burn severity with different sites receiving cameras each year. In 2014 we deployed a total of 53 cameras, 41 of which were placed in the burned area (unburned-very low=13, low=12, moderate=12, high=4) and 12 in the unburned area. In 2015 we deployed a total of 59 cameras with 46 in the burned area (unburned-very low=10, low=14, moderate=16, high=6) and 13 across the unburned area. We placed the majority of cameras in trees focused on the tree hair tube. If there was not an appropriate site to place a camera in a tree we focused it on a feature that could be used by flying squirrels as a launch site or landing platform such as a log or snag. We checked camera traps at the same intervals as the hair tubes, at which time we removed the memory card, and replaced it with a blank card. A camera survey was on the same scale as hair tubes, with a survey being the time between visits to the site. We analyzed photos and recorded any species detected and the date of detection with an emphasis on identifying photos of flying squirrels.

2.4. *Habitat Measurements*--At each random sampling site, we established a 10m

radius plot around the tree bearing the hair tube, in which we measured vegetation and

environmental features. In each plot, we measured the diameter at breast height (DBH;

cm) of all living woody plants and categorized each as shrub (<10 cm DBH), or tree ($\geq$10

cm DBH; Doumas and Koprowski 2013). We recorded the species and number of woody

plants within each category, distance (m), and bearing from the center tree. We assessed

canopy connectivity by counting the number of tree canopies adjoining the canopy of the

center tree (Sterling et al. 2013). We recorded the number (density), genus, and size of

dead trees (logs and snags >2 m long and 20 cm in diameter; Doumas and Koprowski

2013) that fell within, or partially in the plot. At 5 m and 10 m intervals from the center

of the plot we measured canopy density, understory density, litter depth, and duff depth

in each of the four cardinal directions. We use a spherical densiometer and the Strickler

method (Strickler 1959) to measure canopy density. We used a cover pole (2.5 cm x 200

cm) marked in 10 cm increments to measure understory density (Griffith and Youtie

1988). We recorded the highest obscured 10 cm band and recorded any vegetation taller

than the pole as >2 m. We measured duff and litter by excavating a small hole until the

mineral soil layer was reached and measured depth of each layer. We calculated four

additional habitat characteristics in ArcMap using data acquired from United States

Geological Survey (USGS). We used the National Hydrography Dataset (U.S. Geological

survey, 2013, National Hydrography Geodatabase, accessed 25 Jan 2018) and the near

tool in ArcMap to calculate distance to the nearest water course for each site. We also

used the national Digital Elevation Model (U.S. Geological Survey, 2013, USGS NED

n35w118 1/3 arc-second 2013 1x1 degree ArcGrid, accessed 25 Jan 2018) to determine

the elevation for each site. We used ArcMap and the elevation model to calculate the

slope and aspect for each site.

      2.5. *Data Analysis*-- We used the unmarked package (Fiske and Chandler 2011) in

program R (R core team 2017) to determine occupancy and detection rates of flying

squirrels. We used a multi-step approach (Gray and Steidl 2015) to determine if fire

affected flying squirrel occupancy. We created four categories of related occupancy

variables and constructed occupancy models to eliminate extraneous variables. The

terrain category included the variables elevation, slope, aspect, and study area. The forest

floor category included mean litter depth, mean duff depth, and distance to water. The

species composition category included the number of tree species, mean snag class,

percent of trees that were in genus *Quercus*, percent of trees that were conifers, and

percent of trees of other species. The forest structure category included mean DBH, tree

density, canopy connectivity, mean understory density, mean canopy cover, mean log

length, mean log diameter, and log density. We also used the detection covariates days

since the first survey began, the month surveyed, the length of the survey period, if a

camera was present during the survey, and the year the survey occurred (2014 or 2015).

We selected a top model that only contained influential covariates ($P<0.05$ based on a Z

test) for each category. All remaining detection and occupancy covariates were combined

and a final model with only influential covariates was selected ($P<0.05$ based on a Z test).

We added the fire dependent variables of tree mortality and burn severity to this final

model and ranked them using Akaike's Information Criterion (AIC) values.

We also ran separate detection models for camera and hair tubes to determine possible influences of survey method on animal detection. Detection covariates for each set of models varied. Covariates for camera detection models were days since the first survey began, the month surveyed, the length of the survey period, and location of the camera (in a tree or on the ground). Covariates for hair tube detection models were days since the first survey began, the month surveyed, the length of the survey period, and the year the survey occurred (2014 or 2015). We once again ran models for all possible combinations of detection covariates and ranked them according to AIC values, retaining models with a ΔAIC < 2.

3. RESULTS--We detected flying squirrels a total of 357 times at a total of 76 sites. Within each burn severity the total number of detections and the number of sites flying squirrels were detected at decreased as burn severity increased (Fig. 1) In 2014, flying squirrels were detected at 12 sites by both camera traps and hair tubes, at 5 sites by just camera traps, and at 32 sites by just hair tubes. Of the hair tube detections, seven sites had cameras, but six were not focused on the hair tube and one did not have a camera for the session with a hair tube detection. In 2015, using both the hair tubes and cameras 30 sites detected flying squirrels, 6 with just camera detections, and 37 sites with just hair tube detections (none of which were paired with a camera trap).

The top terrain model included the variables elevation, slope, and study area (Table 1). The top forest floor model contained mean duff depth (Table 1). The top forest composition model contained mean snag class (Table 1). The top forest structure model included mean canopy cover and mean log length (Table 1). The top detection model

included the covariates days since the first survey began, the length of the survey period, if a camera was present during the survey, and the year the survey occurred (2014 or 2015). The overall top occupancy model included the occupancy covariates mean canopy cover, mean duff depth, and slope. The detection covariates in the overall top model were days since the first survey began, the length of the survey period, if a camera was present during the survey, and the year the survey occurred (2014 or 2015). Occupancy probability was found to increase with increasing slope, increased mean canopy cover, and deeper duff (Fig. 2). Detection probability increased the more days since the first survey had passed, during longer surveys, when a camera was present, and in 2014. When fire variables were added and ranked using AIC (Table 2) the best fitted model contained just tree mortality, followed by the model containing tree mortality and burn severity, then the model with no burn covariates, and the model containing just burn severity was ranked lowest. Occupancy probability increased at lower tree mortalities (Fig. 2) and burn severities.

Five top detection models from camera data (Table 3) were selected with the top model containing the variables days since the first survey began, the month surveyed, the length of the survey period, and the camera location. Camera location accounted for 100% of model weights, length of the survey period accounted for 81%, days since the first survey accounted for 68%, and the month surveyed in accounted for 66%. We found that detection probabilities increased when cameras were placed in trees versus on the ground, as days since the first survey began increased, the longer the survey period was, and the later in the year the survey happened.

27

Only one top detection model was selected for hair tube data (Table 3). The model showed the variables days since the first survey began, the length of the survey period, and the month surveyed were most important for species detection. This model showed detection probabilities to increase as days since the first survey began increased, the longer the survey period was, and the later in the year the survey occurred.

4. DISCUSSION--Flying squirrels were detected across all burn severities, but most frequently in lower burn severities (1-3). Detections in the high burn severities all occurred at sites closely bordering occupied sites in lower burn severities.

For both cameras and hair tubes, the days since the first survey began, the length of the survey period, and the month surveyed all increased the probability of detecting a flying squirrel. Detection probabilities for hair tubes were also greatly improved by the presence of a trail camera, especially when focused on the tree hair tube. In some cases, debris such as dirt and hair from other species accumulated on the tape in hair tubes making them less effective at collecting hair samples. When a camera was present flying squirrels were still detected by cameras even when a hair sample was not left. Cameras also documented animals approaching hair tubes but not entering them. Days since the first survey likely improved detection as flying squirrels acclimated to the presence of equipment and to identify hair tubes as a food source. Latency in initial flying squirrel detections occurs when camera surveys began (Diggins et al. 2016). Increased length of survey periods allowed more time for the detection of animals. Month likely influenced detection due to the breeding cycle of flying squirrels, with young becoming more

28

independent and dispersing later in the year (Booth 1946). Similar autumnal increases occurred in surveys of other populations of flying squirrels (Boulerice and Fleet 2016).

We found both survey methods to have costs and benefits. Hair tubes are cheaper to produce (Gurnell et al. 2004, Murray 2005) and can be deployed at more sites when funds are limited (Lindenmayer et al. 1999, Sanecki and Green 2005), but require more time to identify species and have a higher chance of species misidentification (Lobert et al. 2001, Gardner et al. 2007). In addition, the minor amount of baiting in hair tubes may result in more exploration by granivores such at flying squirrels. One example of potential misidentification from hair tubes is that dusky footed woodrat (*Neotoma fuscipes*) hair is very similar to flying squirrel hair when viewed under a microscope and extra diligence and the use of scale patterns are needed to correctly identify samples. While cameras allow easier and faster species identification they are more expensive and have a higher rate of failure (e.g. dead batteries, misplacement, etc.; Wellington et al. 2014). While hair tubes on their own were enough to detect flying squirrels, using both sampling methods increased detections and obtained the best results for our surveys. Combining these two methods has proven successful for detecting flying squirrels elsewhere (Boulerice and Fleet 2016).

In the terrain category the variables elevation, slope, and study area were influential. In the San Bernardino Mountains, coniferous forests that flying squirrels occupy only occur at higher (Grinnell 1908, 1933). However, occupancy increased with decreasing elevation. The most likely explanation for increased occupancy at lower elevations comes from the tendency of fire to burn uphill (Rothermel 1972), which would destroy more flying squirrel habitat at higher elevations. The 2009 Grass valley fire

29

started at 1,530 m, close to the mean elevation for the burned study area (1,595±75 m), resulting in more habitat destruction and higher burn severities at higher elevations. This is reflected in the mean elevation for sites with a flying squirrel detection (1,576±71 m) being lower than those without any detections (1,615±74 m). Flying squirrel occupancy was found to increase on steeper slopes. The slope of an area, like elevation, can affect how wildfires burn, with fire being more likely to climb steep slopes and less likely to burn downslope (Rothermel 1972, Weise and Biging 1996). This would result in steeper sites at elevations below where the fire started suffering less habitat damage, leaving more habitat for use by flying squirrels after the fire event. Occupancy probability increased in the unburned study area, likely because this area contains undisturbed habitat.

The occupancy variables included in top models correspond to habitat characteristics in lower burn severities (Parson et al. 2010). In the forest floor category, a deeper duff layer increased occupancy probability. Increased duff depth improves truffle growth that Humboldt's flying squirrels rely on in other parts of their range (Gunther et al. 1983, Maser and Maser 1988, Hall 1991, Meyer et al. 2007). In the forest structure category increased mean canopy cover and longer mean log length both increased occupancy probability. Increased canopy cover can aid in navigating the forest canopy (Mowrey and Zasada 1984) as well as reduce aerial depredation by providing cover and reducing glide distances (Carey 2000, Pyare and Longland 2002). Mean log length could be influenced by areas of more mature forest that squirrels occupy having a taller canopy height resulting in longer logs when trees fall. In the species composition category occupancy probability increased at sites with lower mean snag class (i.e. a greater

number of living trees). Having live trees present increased the available habitat and improves features such as canopy cover and duff depth (Carey 1995).

Tree mortality and burn severity affected occupancy probability of flying squirrels. When the fire variables were added to the top model contain duff depth, canopy cover, and slope model fit improved. Tree mortality appeared in both top ranked models, resulting in increased occupancy probability at lower tree mortalities. Flying squirrels an arboreal species that rely on live trees, so retaining live trees after a fire increase the available habitat. Burn severity affected occupancy to a lesser extent. Burn severity occurred in the second-best model with tree mortality, but the model containing just burn severity was ranked worse than the model containing no fire variables. While burn severity can affect occupancy, the habitat features that higher severity burns affect are not as important compared to the retention of live trees. The Mount Graham red squirrel (*Tamiasciurus hudsonicus grahamensis*) a similarly isolated subspecies can also persist after fires but having live trees present increases this (Koprowski et al. 2006, Leonard and Koprowski 2010). Abert's squirrels (*Sciurus aberti*) are a species that have been introduced to several isolated mountain ranges in the southwest where they were found to inhabit burned areas more often than native species (Gwinn and Koprowski 2017, Ketcham et al. 2017). Abert's squirrels are larger and less specialized (Dodd et al. 1998) than the San Bernardino flying squirrel and Mount Graham red squirrel so can utilize burned areas. Mexican fox squirrels (*Sciurus nayaritensis*) in the Chiricahua Mountains showed a higher use of moderately burned areas compared to low or high burns (Doumas and Koprowski 2012, 2013). These examples all illustrate the varying responses a species can have to fire depending on their niche.

Slope, mean canopy cover, mean duff depth, and tree mortality all appear to have the greatest effect on flying squirrel occupancy. The effect of tree mortality is straight forward as it is a measure of the number of trees killed in an area. Sites with a lower tree mortality contain more living trees after a fire event, increasing the chances of flying squirrel occupancy. Steep areas below where the fire started having lower tree mortality and retaining more habitat. This preserved habitat contains increased canopy cover, which is important flying squirrel populations (Pyare and Longland 2002, Meyer et al. 2007). Canopy cover influences several other habitat features such as increased canopy connectivity (Carey 1991) and heightened truffle availability (Lehmkuhl et al. 2004). In the Cascade Range in Washington no flying squirrel detections were made below a minimum canopy cover threshold of 55% (Lehmkuhl et al. 2006). While our study had detections of flying squirrels with canopy cover as low as 25%, the mean canopy cover for all sites with detections was 68±2%. This intact habitat also contains a deeper duff layer that increases the available truffles for flying squirrels (Gunther et al. 1983, Maser and Maser 1988, Hall 1991, Meyer et al. 2007).

5. CONCLUSIONS--Our study suggests that while fire events can be detrimental to the San Bernardino flying squirrel they can persist if important habitat features, such as an intact forest canopy, deep duff, and live trees remain. This is like other species that can persist in less damaged patches of habitat after a fire event (Bradstock et al. 2005). As climate change increases the likelihood of more intense fire events (Westerling et al. 2006), the San Bernardino flying squirrel may continue to persist in its current but limited range. Our study outlines the need for mature trees with an intact forest canopy to be

retained especially in steeper terrain for the conservation of this subspecies. The ability of this subspecies to persist after fire events at the southernmost limits of the species range illustrates the ability of the species and populations further north to adjust as environmental conditions are affected by climate change

LITERATURE CITED

AITKEN, S. N., S. YEAMAN, J. A. HOLLIDAY, T. WANG, AND S. CURTIS-MCLANE. 2008. Adaptation, migration, or extirpation: climate change outcomes for tree populations. Evolutionary Applications 1:95-111.

BENTZ, J. A., AND P. M. WOODARD. 1988. Vegetation characteristics and bighorn sheep use on burned and unburned areas in Alberta. Wildlife Society Bulletin 16:186–193.

BOHMANN, K., A. EVANS, M. T. P. GILBERT, G. R. CARVALHO, S. CREER, AND M. KNAPP. 2014. Environmental DNA for wildlife biology and biodiversity monitoring. Trends in Ecology and Evolution 29:358–367

BOOTH, E. S. 1946. Notes on the life history of the flying squirrel. Journal of Mammalogy 27:28-30.

BONFILS, C., P. B. DUFFY, B. D. SANTER, T. M. L. WIGLEY, D. B. LOBELL, T. J. PHILLIPS, AND C. DOUTRIAUX. 2008a. Identification of external influences on temperatures in California. Climatic Change 87:S43-S55.

BONFILS, C., B. D. SANTER, D. W. PIERCE, H. G. HIDALGO, G. BALA, T. DAS, T. P. BARNETT, D. CAYAN, C. DOUTRIAUX, A. W. WOOD, A. MIRIN, AND T. NOZAWA. 2008b. Detection and attribution of temperature changes in the mountainous western United States. Journal of Climate 21:6404-6424.

BOULERICE, J.T., AND L.A. VAN FLEET. 2016. A novel technique for detecting Northern flying squirrels. Wildlife Society Bulletin 40:786-791.

BRADSTOCK, R.A., M. BEDWARD, A.M. GILL, AND J.S. COHN. 2005. Which mosaic? A landscape ecological approach for evaluating interactions between fire regimes, habitat, and animals. Wildlife Research 32:409-423.

BRIVIO, F., S. GRINGNOLIA, N. SICA, S. CERISE, AND B. BASSANO. 2015. Assessing the impact of capture on wild animals: The case study of chemical immobilization on alpine ibex. PLoS ONE 10:e0130957.

BROWN, J. K., AND J. K. SMITH, EDS. 2000. Wildland fire in ecosystems: effects of fire on flora. United States Department of Agriculture Forest Service, Rocky Mountain Research Station, Ogden, Utah, RMRS-GTR-42-vol. 2.

BUTLER, R., C. SCHIFFER, AND A. MANN. 1991. Final report--San Bernardino flying squirrel (Conducted under a cooperative agreement with Bear Mtn LTD), United States Department of Agriculture Forest Service, San Bernardino Ranger District Final Report.

CAREY, A. B. 1991. The biology of arboreal rodents in Douglas-fir forests. United States Department of Agriculture Forest Service General Technical Report, Pacific Northwest Research Station, PNW-GTR-276.

CAREY, A. B. 1995. Sciurids in Pacific Northwest managed and old-growth forests. Ecological Applications 5:648-661.

CAREY, A. B. 2000. Effects of new forest management strategies on squirrel populations. Ecological Applications 10:248-257.

CATTET, M., J. BOULANGER, G. STENHOUSE, R. A. POWELL, AND M. J. REYNOLDS-HOGLAND. 2008. An evaluation of long term capture effect in ursids: Implications for wildlife welfare and research. Journal of Mammalogy 89:973-990.

CAYAN, D., E. P. MAURER, M. D. DETTINGER, M. TYREE, AND K. HAYHOE. 2008. Climate change scenarios for the California region. Climatic Change 87:S21-S42.

CZECH, B., P. R. KRAUSMAN, AND P. K. DEVERS. 2000. Economic associations among causes of species endangerment in the United States. Bioscience 50:593–601.

DE MARINIS, A. M., AND A. ASPREA. 2006. Hair identification key of wild and domestic ungulates from southern Europe. Wildlife Biology 12:305–320.

DIGGINS, C.A., L. MICHELLE GILLEY, C.A. KELLY, AND W. MARK FORD. 2016. Comparison of survey techniques on detection of northern flying squirrels. Wildlife Society Bulletin 40:644-662.

DODD, N.L., S.S. ROSENSTOCK, C.R. MILLER, R.E. SCHWEINSBURG. 1998. Tassel-eared squirrel population dynamics in Arizona: Index techniques and relationships to habitat condition. Research Branch Technical Report #27, Arizona Game and Fish Department, Phoenix

DOUMAS, S. L., AND J. L. KOPROWSKI. 2012. Effect of heterogeneity in burn severity of Mexican fox squirrels following the return of fire. International Journal of Wildland Fire 22:405-413.

DOUMAS, S. L., AND J. L. KOPROWSKI. 2013. Return of fire as a restoration tool: long term effects of burn severity on habitat use by Mexican fox squirrels. Restoration Ecology 21:133-139.

DUCEY, M.J., W.K MOSER, AND P.M.S. ASHTON. 1996. Effect of fire intensity on understory composition and diversity in Kalmia-dominated oak forest, New England USA. Vegetatio 123:81-90.

FEDERAL REGISTER. 2016. Proposed Rules: San Bernardino flying squirrel (*Glaucomys sabrinus californicus*). Federal Register 81:19534–19538.

FISKE I. AND R. CHANDLER. 2011. unmarked: an R package for fitting hierarchical models of wildlife occurrence and abundance. Journal of Statistical Software 43:1–23.

FISHER, J. T., AND L. WILKINSON. 2005. The response of mammals to forest fire and timber harvest in the North American boreal forest. Mammal Review 35:51–81.

FORAN, D. R., S. C. MINTA, AND K. S. HEINEMEYER. 1997. DNA-based analysis of hair to identify species and individuals for population research and monitoring. Wildlife Society Bulletin 25:840–847.

GARDEN, J. G., C. A. MCALPINE, H. P. POSSINGHAM, AND D. N. JONES. 2007. Using multiple survey methods to detect terrestrial reptiles and mammals: what are the most successful and cost-efficient combinations? Wildlife Research 34:218–227.

GIGLIOTTI, L. C., D. R. DIEFENBACH, AND M. J. SHERIFF. 2017. Geographic variation in winter adaptations of snowshoe hares (*Lepus americanus*). Canadian Journal of Zoology 95:539-545.

GRAY, K.M., AND R.J. STEIDL. 2015. A plant invasion affects condition but not density or population structure of a vulnerable reptile. Biological Invasions 17:1979-1988.

GRIFFITH, B., AND B. A. YOUTIE. 1988. Two devices for estimating foliage density and deer hiding cover. Wildlife Society Bulletin 16:206–210.

GRINNELL, J. 1908. The biota of the San Bernardino Mountains. University of California Publications in Zoology 5:1-170.

GRINNELL, J. 1933. Review of the recent mammal fauna of California. University of California Publications in Zoology 40:71-284.

GURNELL J, P. W. W. LURZ, M. D. F. SHIRLEY, S. CARTMEL, P. J. GARSON, L. MAGRIS, J. STEELE. 2004. Monitoring red squirrel *Sciurus vulgaris* and grey squirrel *Sciurus carolinensis* in Britain. Mammal Review 34:51–74.

GUNTHER, P. M., B. S. HORN, AND G. D. BABB. 1983. Small mammal populations and food selection in relation to timber harvest practices in the western Cascade Mountains. Northwest Science 57:32-44.

GWINN, R. N., AND J. L. KOPROWSKI. 2017. Differential response to fire by an introduced and an endemic species complicates endangered species conservation. Hystrix 27:1-7.

HALL, E. R. 1981. The mammals of North America. John Wiley & Sons, New York, NY.

HALL, D. S. 1991. Diet of the northern flying squirrel at Sagehen Creek, California. Journal of Mammalogy 72:615-617.

HAMPE, A. AND R. J. PETIT. 2005. Conserving biodiversity under climate change: the rear edge matters. Ecology Letters 8:461-467.

HARRIS, R. B., J. WINNIE, S. J. AMISH, A. BEJA-PEREIRA, R. GODINHO, V. COSTA, AND G. LUIKART. 2010. Argali abundance in the Afghan Pamir using capture-recapture modeling from fecal DNA. Journal of Wildlife Management 74:668-677.

HAYHOE, K., D. CAYAN, C. B. FIELD, P. C. FRUMHOFF, E. P. MAURER, N. L. MILLER, S. C. MOSER, S. H. SCHNEIDER, K. N. CAHILL, E. E. CLELAND, L. DALE, R. DRAPEK, R. M. HANEMANN, L. S. KALKSTEIN, J. LENIHAN, C. K. LUNCH, R. P. NEILSON, S. C. SHERIDAN, AND J. H. VERVILLE. 2004. Emissions pathways, climate change, and impacts on California. Proceedings of the National Academy of Sciences 101:12422-12427.

HOLMES, K. A., K. E. VEBLEN, T. P. YOUNG, AND A. M. BERRY. 2008. California oaks and fire: A review and case study. General Technical Report, edited by A. Merenlender, D. McCreary, and K.L. Purcell. United States Forest Service, Albany, CA: Pacific Southwest Research Station, PSW-GTR-217.

KELLY, A. E., AND M. L. GOULDEN. 2008. Rapid shifts in plant distribution with recent climate change. Proceedings of the National Academy of Sciences of the United States of America 105:11823-11826.

KETCHAM, S. L., J. L. KOPROWSKI, D. A. FALK. 2017. Differential response of native Arizona gray squirrels and introduced Abert's squirrels to a mosaic of burn severities. Mammal Study 42:247-258.

KOPROWSKI, J. L., K. M. LEONARD, C. A. ZUGMEYER, AND J. L. JOLLEY. 2006. Direct effects of fire on endangered Mount Graham red squirrels. Southwestern Naturalist 51:59–63.

LAVOIE, M., G. STARR, M.C. MACK, T.A. MARTIN, AND H.L. GHOLZ. 2010. Effects of a prescribed fire on understory vegetation, carbon pools, and soil nutrients in longleaf pine-slash pine forest in Florida. Natural Areas Journal 30:82-94.

LEHMKUHL, J. F. 2004. Epiphytic lichen diversity and biomass in low-elevation forest of the eastern Washington Cascade range, USA. Forest Ecology and Management 187:381-392.

LEHMKUHL, J. F., K. D. KISTLER, J. S. BEGLEY, AND J. BOULANGER. 2006. Demography of northern flying squirrels informs ecosystem management of western interior forests. Ecological Applications 16:584-600.

LEONARD, K. M., AND J. L. KOPROWSKI. 2010. Effects of fire on endangered Mount Graham red squirrels (*Tamiasciurus hudsonicus grahamensis*): responses of individuals with known fates. Southwestern Naturalist 55:217-224.

LINDENMAYER, D. B., R. D. INCOLL, R. B. CUNNINGHAM, M. L. POPE, C. F. DONNELLY, C. I. MACGREGOR, C. TRIBOLET, AND B. E. TRIGGS. 1999. Comparison of hair tube types for the detection of mammals. Wildlife Research 26:745-753.

LOBERT, B., L. LUMSDEN, H. BRUNNER, AND B. TRIGGS. 2001. An assessment of the accuracy and reliability of hair identification of south-east Australian mammals. Wildlife Research 28:637-641.

LUKACS, P. M., AND K. P. BURNHAM. 2005. Review of capture-recapture methods applicable to non-invasive genetic sampling. Molecular Ecology 14:3909-3919.

MASER, C. AND Z. MASER. 1988. Interactions among squirrels, mycorrhizal fungi, and coniferous forests in Oregon. Great Basin Naturalist 48:358-369.

MEYER, M. D., M. P. NORTH, AND D. A. KELT. 2005. Fungi in the diets of northern flying squirrels and lodgepole chipmunks in the Sierra Nevada. Canadian Journal of Zoology 83:1581-1589.

MEYER, M. D., D. A. KELT, AND M. P. NORTH. 2007. Microhabitat associations of northern flying squirrels in burned and thinned forest stands of the Sierra Nevada. American Midland Naturalist 157:202-211.

MONDOL, S., K. U. KARANTH, N. S. KUMAR, A. M. GOPALASWAMY, A. ANDHERIA, AND U. RAMAKRISHNAN. 2009. Evaluation of non-invasive genetic sampling methods for estimating tiger population size. Biological Conservation 142:2350–2360.

MOWREY, R. A., AND J. C. ZASADA. 1984. Den tree use and movements of northern flying squirrels in interior Alaska and implications for forest management. Pg. 351-356. In: W.R. Meehan, T.R. Merrell Jr, and T.A. Hanley (eds.) Fish and wildlife relationships in old growth forests: proceedings of a symposium, Juneau, Alaska, Apr. 1982. American Institute of Fishery Research Biologists, Morehead City, North Carolina.

MULCAY, D. M. 2003. Surgical implantation of transmitters into fish. Institute for Laboratory Animal Research Journal 44:295-306.

MURRAY, A. J. 2005. A new low-cost hair tube design for the detection of the spotted-tailed quoll Dasyurus maculatus in south-eastern Australia. Australian Mammalogy 27:81–84.

NAKASHIMA, Y., K. FUKASWA, AND H. SAMEHIMA. 2018 Estimating animal density without individual recognition using information derivable exclusively from camera traps. Journal of Applied Ecology 55:735-744.

PATTEN, M. A., AND J. C. BURGER. 2018. Reserves as double-edged sword: Avoidance behavior in an urban-adjacent wildland. Biological Conservation 218:233-239.

PARMESAN, C., AND G. YOHE. 2003. A globally coherent fingerprint of climate change impacts across natural systems. Nature 421:37-42.

PARMESAN, C. 2006. Ecological and evolutionary responses to recent climate change. Annual Review of Ecology Evolution and Systematics 37:637-669.

PARSON, A., P. R. ROBICHAUD, S. A. LEWIS, C. NAPPER, AND J. T. CLARK. 2010. Field guide for mapping post-fire soil burn severity. U.S. Department of Agriculture Forest Service Fort Collins, CO, Rocky Mountain Research Station, General Technical Report RMRS-GTR-243:49.

PEARSON, G. A., A. LAGO-LESTON, AND C. MOTA. 2009. Frayed at the edges: Selective pressure and adaptive response to abiotic stressors are mismatched in low diversity edge populations. Journal of Ecology 97:450-462.

PYARE, S., AND W. S. LONGLAND. 2002. Interrelationships among northern flying squirrels, truffles, and microhabitat structure in Sierra Nevada old-growth habitat. Canadian Journal of Forest Research 32:1016-1024.

ROTHERMEL, R. C. 1972. A mathematical model for predicting fire spread in wildland fuels. U.S. Department of Agriculture, Intermountain Forest and Range Experiment Station, Ogden, Utah, RS INT-115:40.

RUSSELL, W. M. S., AND R. L. BURCH. 1959. The principles of humane experimental
   technique. Methuen and Co. Ltd., London, UK.

SANECKI, G.M, AND K. GREEN. 2005. A technique for using hair tubes beneath the
   snowpack to detect winter-active mammals in the subnivean space. European
   Journal of Wildlife Research 51:41-47.

SCOTTS, D. J. AND S. A. CRAIG. 1988. Improved hair-sampling tube for the detection of
   rare mammals. Wildlife Research 15:469-472.

SMITH, W. P. 2007. Ecology of *Glaucomys sabrinus*: habitat, demography, and
   community relations. Journal of Mammalogy 88:862-881.

STEPHENSON, J. R., AND G. M. CALCARONE. 1999. Southern California mountains and
   foothills assessment: habitat and species conservation issues. United States
   Department of Agriculture Forest Service, Albany, CA: Pacific Southwest
   Research Station, General Technical Report GTRPSW-175:402.

STERLING, E., N. BYNUM, AND M. BLAIR. 2013. Primate Ecology and Conservation. Page
   88. Oxford University Press, Oxford, United Kingdom.

STRICKLER, G. S. 1959. Use of the densiometer to estimate density of forest canopy on
   permanent sample plots. United States Department of Agriculture Forest Service,
   Pacific Northwest Forest and Range Exp. Sta. Research Note 180, Portland,
   Oregon, PNW-RN180:5.

SUCKLING, G. C. 1978. A hair sampling tube for the detection of small mammals in trees.
   Wildlife Research 5:249-252.

TAYLOR, R. J. 1985. Identification of the hair of Tasmanian mammals. Papers and
   Proceedings of the Royal Society of Tasmania 119:69–82.

TEERINK, B.J. 1991. Hair of West European Mammals: Atlas and Identification Key.

  Cambridge University Press, Cambridge, UK.

THOMSEN P. F., AND E. WILLERSLEV. 2015. Environmental DNA–An emerging tool in

  conservation for monitoring past and present biodiversity. Biological

  Conservation 183:4–18.

TRENBERTH, K. E., P. D. JONES, P. AMBENJE, R. BOGARIU, D. EASTERLING, A. KLEIN

  TANK, D. PARKER, F. RAHIMZADEH, J. A. RENWICK, M. RUSTICUCCI, B. SODEN,

  AND P. ZHAI. 2007. Observations: Surface and atmospheric climate change in

  Climate change 2007: The physical science basis. Contribution of working group

  I to the fourth assessment report of the intergovernmental panel on climate

  change. (S. SOLOMON, D. QIN, M. MANNING, Z. CHEN, M. MARQUIS, K. B.

  AVERYT, M. TIGNOR, AND H. L. MILLER, eds.). Cambridge University Press,

  Cambridge, United Kingdom, and New York, NY, USA.

U.S. GLOBAL CHANGE RESEARCH PROGRAM. 2009. Global climate change impacts in the

  United States. U.S. Global Change Research Program. Thomas R. Karl, Jerry M.

  Melillo, and Thomas C. Peterson, (eds.). Cambridge University Press, 2009.

WATERS, J. R., AND C. J. ZABEL. 1995. Northern flying squirrel densities in fir forests of

  northeastern California. Journal of Wildlife Management 59:858-866.

WEISE, D.R., G.S. BIGING, 1996. Effects of wind velocity and slope on flame properties.

  Canadian Journal for Forest Research26:1849–1858.

WELLINGTON, K., C. BOTTOM, C. MERRIL, AND J.A. LITVAITIS. 2014. Identifying

  performance differences among trail cameras used to monitor forest mammals.

  Wildlife Society Bulletin 38:634-638.

WESTERLING, A. L., H. G. HIDALGO, D. R. CAYAN, AND T. W. SWETNAM. 2006. Warming

and earlier spring increase western US forest wildfire activity. Science 313:940–

943.

YOON, J, S. WANG, R. R. GILLIES, L. HIPPS, B. KRAVIT, AND P. J. RASCH. 2015. Extreme

fire season in California: A glimpse into the future. Bulletin of the American

Meteorological Society 96:S1-5.

TABLE 1—The effect of variables on San Bernardino flying squirrel occupancy or detection. The shown p-value is from best fitted model the variable occurred in. DBH=Diameter at Breast Height, SE=Standard Error.

Category	Variable	Estimate	SE	Z value	p-value
Terrain	Intercept	8.14	1.49	5.47	<0.01
	Elevation (m)	<-0.01	<0.01	-5.60	<0.01
	Slope (°)	0.09	0.03	3.67	<0.01
	Burned Study Area	-1.29	0.48	-2.69	<0.01
	Aspect (°)	<0.01	<0.01	0.05	0.90
Forest Floor	Intercept	-1.01	0.32	-3.13	<0.01
	Mean Duff Depth (cm)	0.16	0.04	3.94	<0.01
	Distance to Water (m)	<-0.01	<0.01	-1.52	0.13
	Mean Litter Depth (cm)	0.09	1.24	0.73	0.47
Species Composition	Intercept	0.49	0.23	2.20	0.03
	Mean Snag Class	-0.93	0.36	-2.56	0.01
	Number of Species	0.28	0.16	1.72	0.08
	% Quercus	-0.68	0.63	-1.07	0.28
	% Other	-0.72	0.98	-0.74	0.46
	% Evergreen	-9.67	13.30	-0.73	0.47
Forest Structure	Intercept	-3.56	0.67	-5.33	<0.01
	Mean Canopy Cover (%)	0.06	<0.01	5.65	<0.01
	Mean Log Length (m)	0.10	0.03	2.54	0.01
	Mean Understory Density (cm)	<-0.01	0.01	-0.79	0.42
	Mean DBH	-0.01	0.02	-0.60	0.55
	Mean Log Diameter (cm)	-0.01	0.02	-0.59	0.55
	Tree Density	-0.02	0.04	-0.47	0.63
	Canopy Connectivity	0.05	0.10	0.47	0.63
	Log Density	-0.03	0.11	-0.26	0.80
Detection	Intercept	0.52	0.49	1.07	0.20
	Survey Year	<-0.01	<0.01	-8.08	<0.01
	Days Since 1st Survey	<0.01	<0.01	7.72	<0.01
	Camera Present	1.28	0.19	6.67	<0.01
	Survey Length	0.04	0.01	3.63	<0.01
	Survey Month	0.11	0.07	1.67	0.09

TABLE 2- Ranking of top models for explaining San Bernardino flying squirrel occupancy with and without fire occupancy variables. Detection probability was the same for all models (0.51±0.04). Detection model parameters are as follows: days=number of days since first survey, plength=length of survey, cam=if a camera was present at a site, and year=year surveyed. Occupancy model parameters are as follows: mortality=tree mortality, severity=burn severity, slope=slope, canopy cover= mean canopy cover, and duff=mean duff depth.

Model	K^a	AIC^b	ΔAIC^c	w_i^d	Ψ^e	ΨSE^f
p (days + plength + cam + year) Ψ (mortality + slope + canopy cover + duff)	10	954.84	0.00	0.72	0.53	0.07
p (days + plength + cam+ year) Ψ (mortality + severity + slope + canopy cover + duff)	11	956.85	2.01	0.26	0.52	0.08
p (days + plength + cam + year) Ψ (slope + canopy cover + duff)	9	962.66	7.83	0.01	0.52	0.07
p (days + plength + cam + year) Ψ (severity + slope + canopy cover + duff)	10	964.06	9.22	<0.01	0.54	0.07

[a]K=number of parameters
[b]AIC= Akaike's Information Criterion
[c]Δ AIC= AIC relative to the most parsimonious model
[d]w_i=AIC model weights
[e]Ψ =occupancy probability
[f]Ψ SE=standard error for occupancy probability

TABLE 3--Top ranking models (Δ AIC <2.00) estimating explanatory variables for two survey methods (cameras and hair tubes) looking at the effects of survey features on detection probability of the San Bernardino flying squirrel in the San Bernardino Mountains in southern California, USA, 2014-2015. Results are based on the intercept model for occupancy: Ψ (.). Model parameters are as follows: days=number of days since first survey, month=the month surveyed, plength=length of survey, location=location of camera (tree or ground).

Model	K^a	AIC^b	ΔAIC^c	w_i^d
Camera detection models				
p(days + month + location + plength)	6	339.84	0.00	0.31
p(days + month + location)	5	340.84	0.99	0.19
p(days + location + plength)	5	340.88	1.03	0.18
p(month + location + plength)	5	341.13	1.29	0.16
p(location + plength)	4	341.24	1.39	0.15
Hair tube detection model				
p(days + month + plength)	5	983.01	0.00	1

[a]K=number of parameters
[b]AIC=Akaike's Information Criterion
[c]Δ AIC=AIC relative to the most parsimonious model
[d]w_i=AIC model weights

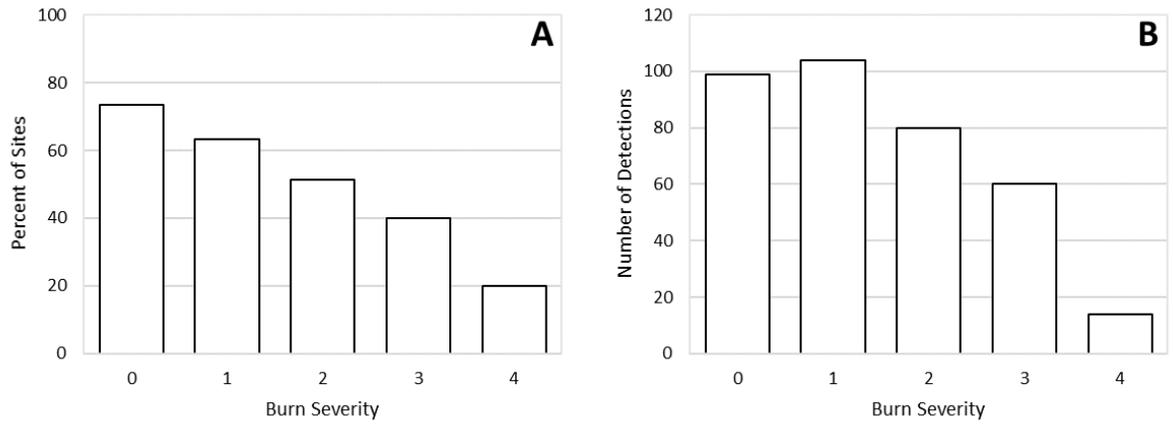Fig. 1--The percentage of sites with San Bernardino flying squirrel detections (A) and the total number of San Bernardino flying squirrel detections (B) across all surveyed burn severities for surveys in the San Bernardino Mountains in 2014 and 2015. Burn severity 0 represents the unburned study area with the remaining four burn severities located in the burned study area: unburned-very low (1), low (2), moderate (3), and severe (4).
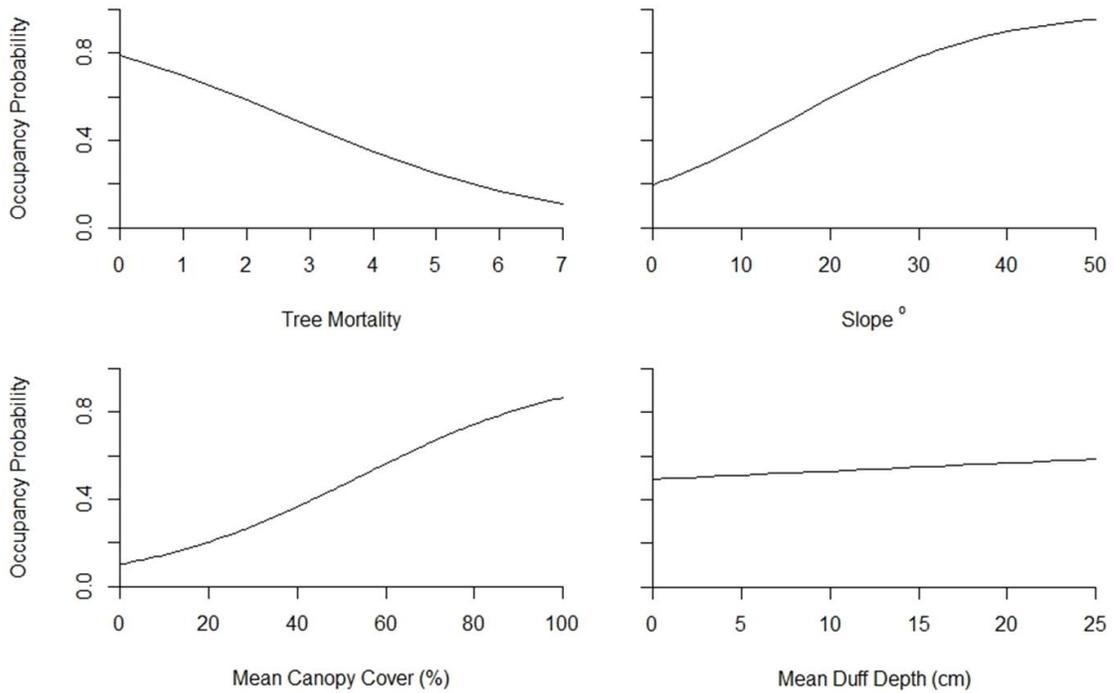
FIG. 2--Effect of mean canopy cover, mean duff, slope, and tree mortality on occupancy of the San Bernardino flying squirrel in the San Bernardino Mountains in southern California, USA, 2014-2015. Results are based on the top ranked model: p (days + survey length + camera + year) Ψ (mean duff depth + slope + mean canopy cover+ tree mortality).

APPENDIX B: A COMPARISON OF NON-INVASIVE METHODS FOR SAMPLING
SPECIES RICHNESS IN A FIRE AFFECTED LANDSCAPE

Maxwell N. Mazzella and John L. Koprowski

A Comparison of Non-invasive Methods for Sampling Species Richness in a Fire
Affected Landscape

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Abstract

Background

Wildfire can drastically alter landscapes and create a mosaic of burn severities. These varied landscapes alter habitat which can affect species occupancy. Mammals are a diverse taxon with varying, species-specific reactions to fire events. We surveyed the mammalian species present after a fire event in the San Bernardino Mountains of southern California. We used and compared the efficacy of two non-invasive sampling methods, camera traps and hair tubes, to determine the occupancy of mammals in a post fire landscape.

Results

We detected a total of 14 species across areas of differing burn severities and tree mortality. Ten species were detected across all burn types, but four species were only detected at lower severities. Nine species were detected across all tree mortalities, but five were only detected at lower mortalities. Camera traps detected 14 species and hair tubes detected 8 species. Species richness decreased as burn severity and tree mortality increased.

Conclusions

Large bodied and generalist species can better utilize all areas affected by fires. More specialized species are still able to exist in a post fire landscape as long as key habitat features remain intact. Cameras detected more species overall, but a relatively higher failure rate and monetary cost can limit their use. Hair tubes were more monetarily

cost-effective and readily detected certain species, though time spent processing and identifying samples was much greater. These two methods seem to work best in tandem, as they can offset many of the issues of the other.

Background

      Wildfire is an integral process in many ecosystems worldwide, but in a growing number of places how and when fire events occur is changing. The southwest United States has had a drastic increase in wildfire events in recent history (Dennison et al. 2014, Owen et al. 2012). Previously, wildfires burned at low intensities, over small areas, and at low frequencies (Brown and Smith 2000). Fire events have become more numerous and severe due to several factors including anthropogenic fire suppression, which has led to an increase in accrued woody fuels that propagate wildfire (Westerling et al. 2006), and ongoing drought conditions driven by climate change (Westerling et al. 2006). Fires of greater intensity result in larger, more uniformly burned areas as opposed to a mosaic created by infrequent, low intensity fire (Bonnicksen and Stone 1982, Halofsky et al. 2011). Such changes in fire regimes can lead to severe consequences for forest dwelling wildlife, including decreased available habitat (Fisher and Wilkinson 2005) and increased mortality (Koprowski et al. 2006).

      Mammals are a diverse taxon that can react differently to wildfire. Small mammals, depending on their habitat requirements, can benefit from or be hindered by a

fire event. Small mammals typically occupy small home ranges and exhibit rapid

reproduction, which allows their response to be quickly assessed after a fire event (Fisher

and Wilkinson 2005). Forest specialists such as flying squirrels (*Glaucomys spp.*) can be

negatively affected by fire (Roberts et al. 2015, Smith 2007), whereas other species like

golden mantled ground squirrels (*Callospermophilus lateralis*) and California ground

squirrels (*Otospermophilus beecheyi*) inhabit intensely burned areas (Converse et al.

2006). Large mammals occupy larger home ranges than small mammals and can

therefore more easily cope with some effects of fire. For example, fires can improve

forage quality for ungulate species (Hobbs and Spowart 1984). Fires can affect

populations of animals differently as well. Species in isolated mountain ranges can be

drastically impacted by disturbance events such as fire due to already limited habitat

being further reduced and difficulty in repopulating metapopulations (Fahrig and

Merriam 1985, 1994, Stacey and Taper 1992).

The San Bernardino Mountains in southern California are an isolated mountain

range that has experienced decreased precipitation and increased temperatures that have

amplified the length and severity of wildfire season. This range is also home to several

mammalian species, including a unique subspecies of the Humboldt's flying squirrel

(*Glaucomys oregonensis*), the San Bernardino flying squirrel (*G. oregonensis*

californicus). Species like the Humboldt's flying squirrel (*Glaucomys oregonensis*) that

rely on mature forests are negatively affected by fire and climate change due to the

alteration and long recovery time of forest ecosystems (Carey 1995, Lehmkuhl et al.

2006, Smith 2007, Waters and Zabel 1995). Climate change is likely to alter the habitat

available to mammals in the San Bernardino Mountains (U.S. Global Change Research

Program 2009) and as conditions become more arid, fire frequency and intensity are expected to increase (Yoon et al. 2015).

Traditional mammalian survey methods usually consist of live-trapping or direct observation, which can be effective and vital to wildlife science, however, they have several drawbacks. Trapping can result in stress to animals, and individuals may become trap-shy after an initial encounter, making subsequent captures more difficult (Mulcay 2003, Cattet et al. 2008). Direct observation can stress animals not used to human presence, and in some cases, animals might avoid surveyors (Mulcay 2003, Cattet et al. 2008), and thereby bias survey results. Non-invasive methods of sampling, such as camera traps and hair tubes, can offset some of these disadvantages. Camera traps can survey long-term and with minimal human presence (Nakashima et al. 2018, Patten and Burger 2018). Hair tubes and other hair snares can easily collect hair or genetic samples that would otherwise require handling of animals (Taylor 1985, Teerink 1991, De Marinis and Asprea 2006). Each method has its own strengths and weaknesses, depending on circumstances (e.g. surveying cryptic, sensitive, or easily stressed species), so knowing which technique to use can improve efficacy and quality of data collected.

The goal of our study was to use non-invasive methods to assess the effects of fire on the mammalian community in the San Bernardino Mountains. We surveyed an area recently affected by a wildfire event as well as a nearby area not impacted by fire to compare mammalian species occupancy. We also assessed the efficacy of two non-invasive sampling methods, hair tubes and camera traps, in detecting various mammals.

Methods

Study Area

Our study area was located near Lake Arrowhead in the San Bernardino Mountains of southern California, USA, at 3500m in elevation. The base of the range is characterized by desert scrub, mid elevations are oak (*Quercus*) woodlands, and the higher elevations are composed of mixed oak-conifer forest. Forests occur at higher elevations due to increased precipitation, primarily from November to March, compared to the surrounding valleys. Despite differences in precipitation along an elevational gradient, wildfire events are common throughout this mountain range.

Our study assessed mammalian occupancy across two focal areas in the San Bernardino Mountains: a recently burned area and a nearby unburned area. The burned area was comprised of 377 ha that burned in 2007 Grass Valley fire. Prior to the fire much of the area was comprised of mature mixed oak-conifer forest. Post fire the area is characterized by a mosaic of vegetation ranging from unburned sites containing mature forest, to high burn severity sites which are open and contain short, shrubby vegetation. We assigned USDA burn severity classifications (Parson et al. 2010) across the burned area, which ranged from 1-4. Category 1 (unburned-very low) is described by almost all surface organics (soil, woody debris, surface roots, etc.) remaining intact with little to no charring, and with vegetation remaining green. Category 2 (low) consists of most surface organics remaining recognizable with light black or brown charring, and vegetation remaining green. In Category 3 (moderate), up to 80% of surface organics are consumed leaving black charring, gray ash, and brown, scorched vegetation. Category 4 (high) is the most severe, with almost all surface organics destroyed, heavy charring and ash, and

almost no vegetation remaining. Tree mortality, which refers to amount of woody

vegetation killed by a fire, across the burned area was assessed and placed into categories

from 1-7 and were as follows: 1= 0%, 2= 1-10%, 3= 10-25%, 4= 25-50%, 5= 50-75%, 6=

75-90%, 7= >90%. Our second focal area, referred to as the unburned area, was in a

nearby tract of unburned forest, where burn severity and tree mortality were classified as

0. We used ArcMap (2015, v10.3.1, ESRI, Redlands, California) to create a total of 145

stratified, random sites across all burn severities at which we surveyed mammals. We

established 115 sites in the burned area across all burn severities (30 unburned-very low,

35 low burn, 35 moderate burn, 15 high burn) and 30 in the unburned area.

Surveys

We surveyed four times from June to October in 2014 and five times from May to

October in 2015. We used hair tubes and trail cameras (Bushnell 8 MP Trophy camera,

model no. 119436C, Bushnell Corporation, Overland Park, Kansas) to survey. Each hair

tube consisted of a 30 cm long x 10 cm diameter PVC pipe that contained duct tape (3M

Company, Maplewood, Minnesota) placed inside on a removable wooden block on top of

each entrance to act as a hair snare. Hair tubes were baited with peanuts and peanut butter

placed in a hardware cloth bait basket secured by a screw in the middle of the hair tube to

retain bait longer. We placed two hair tubes (Suckling 1978, Scotts and Craig 1988,

Lindenmayer 1999) at each random site across both study areas: one in a tree 1-5 m

above the ground, and a second at the base of the tree.

We also placed trail cameras at a random subset of sites in addition to hair tubes.

We placed a total of 53 cameras in 2014: 41 in the burned area (unburned-very low= 13,

low= 12, moderate= 12, high= 4) and 12 in the unburned area. We placed 59 cameras in 2015: 46 in the burned area (unburned-very low= 10, low= 14, moderate= 16, high= 6) and 13 in the unburned area. We primarily focused cameras on the hair tubes located on trees, but in some cases focused cameras on nearby structures such as snags or logs. A survey for both hair tubes and cameras was the amount of time between visits to a site. We visited sampling sites a mean of every 30.8 ($\pm$8.3) days on average with a total mean sampling effort of 123.0 ($\pm$9.0) days in 2014 and 157.8 ($\pm$13.4) days in 2015 per site. Site visits consisted of switching out memory cards from cameras, removing any tape containing hair from hair tubes for later identification, and replacing tape and bait in hair tubes.

Species Identification

We separated hair samples from tape in a laboratory by soaking tape in a xylene bath (Crown, Packaging Service Co., Inc., Pearland, Texas). We placed hair samples on microscope slides, then viewed hair at 40x magnification under a compound light microscope (Nikon Eclipse E200, Nikon Corporation, Minato, Tokyo, Japan). We compared internal structures, the cortex and medulla, as well as external scale patterns of hair samples to reference samples obtained from the University of Arizona mammal collection to determine the species of origin. In addition to hair analysis, we viewed photos from camera traps and recorded any detections of mammals. The number of detections of each species for each survey method was recorded.

Once all samples were identified, species richness was calculated from detections for each survey method (camera and hair tube) and all detections from both methods for

each burn severity and tree mortality. The number of detections of each species for each survey method was also recorded.

Results

Fourteen mammalian species were detected in our surveys (Table 1); black bear (*Ursus americanus*), bobcat (*Lynx rufus*), California ground squirrel *(Otospermophilus beecheyi)*, chipmunk (*Tamias spp.*), coyote (*Canis latrans*), dusky footed woodrat (*Neotoma fuscipes*), gray fox (*Urocyon cinereoargenteus*), mice (*Chaetodipus spp.*, *Reithrodontomys spp.*, and *Peromyscus spp.*), mountain lion (*Puma concolor*), mule deer (*Odocoileus hemionus*), raccoon (*Procyon lotor*), San Bernardino flying squirrel (*Glaucomys oregonensis californicus*), skunk (*Mephitis mephitis*, *M. macroura*, and *Spilogale gracilis*), and Western gray squirrel (*Sciurus griseus*). Two species of chipmunk, *T. speciosus* and *T. merriami*, occur in the surveyed areas, but were indistinguishable and combined into the chipmunk group. Several species of mice belonging to the genera *Chaetodipus*, *Reithrodontomys*, and *Peromyscus* occur in the surveyed areas, but were combined into the mouse group. Three species of skunk, *Mephitis mephitis*, *M. macroura*, and *Spilogale gracilis,* occur in the surveyed areas, but were combined into the skunk group. Cameras detected all 14 species at least once and 8 of the 14 species were detected by hair tubes (Table 1). The six species not detected by hair tubes were bobcats, coyotes, mice, mountain lions, mule deer, and raccoons. Of the detected species nine were detected over 30 times by at least one method across all surveys (Figure 2). These nine species were; black bears, California ground squirrels,

chipmunks, dusky footed woodrats, gray foxes, mice, mule deer, San Bernardino flying squirrels, and Western gray squirrels.

Sites with low burn severities and low tree mortality had the greatest species richness (Figure 1). Ten species were detected in all burn severities (0-4) with bobcats, coyotes, mountain lions, and raccoons being detected only at lower severities (Table 2). Nine species were detected across all tree mortalities (0-7) with bobcats, coyotes, mountain lions, raccoons, and flying squirrels only being detected at lower mortalities (Table 2). Of the nine species that were detected over 30 times all were detected at least once in all burn severities (Figure 2). Seven of the nine species were detected across all tree mortalities except for flying squirrels, which occurred from 0-4 (Figure 2).

Discussion

Cameras detected the 14 species identified in our study at least once, whereas only eight species were detected by hair tubes. Most species detected by hair tubes were small bodied rodents that could easily access hair tubes and were also attracted by bait. Black bears, though large bodied, were attracted to hair tubes by bait (Bongo et al. 2005) and had the dexterity and strength to access it, so were detected by hair tubes regularly. Foxes and skunks were also likely attracted to bait in hair tubes, but could not easily access it, so did not treat hair tubes as food sources and did not visit regularly. Six species detected by cameras were not detected by hair tubes for several reasons. Mice were regularly detected by camera entering hair tubes, but their small size allowed them to avoid contact with tape to leave hair samples. Other studies have dealt with this problem by deploying different sized hair tubes to detect smaller species (Pickerell et al. 2014).

Deer are large and were not able to access bait in hair tubes, thus any photos taken were incidentally. Mountain lions are apex predators that occur in low densities and were not attracted to the bait we used (Choate et al. 2006), so camera detections were incidental. Bobcats, while slightly more numerous, were also detected by camera incidentally, as they were not attracted to hair tube bait. Raccoons had the potential to be attracted to hair tubes but were only detected by cameras three times. The most likely explanation for this is their penchant for riparian and urban areas, which means they are only transient visitors to mixed conifer, oak woodlands (Lutze and Anderson 1979). Coyotes are not often found in dense woodlands (Bekoff 1977).

Species richness was found to be highest at lower burn severities. Ten species were detected across all burn severities: bears, foxes, deer, skunks, ground squirrels, chipmunks, woodrats, mice, gray squirrels, and flying squirrels. The larger bodied species detected (bears, foxes, deer, and skunks) have large home range sizes, so can easily traverse across multiple burn severities. A camera trapping survey in the San Bernardino Mountains near Fawnskin, CA captured several large predators detected in our study, including black bears, bobcats, and gray foxes, utilized both burned and unburned sites, except for gray foxes which were more commonly detected in burned areas (Borchert 2012). Black bears elsewhere also use burned areas, but favor lower severities (Cunningham et al. 2003, Stratman and Pelton 2007). Gray foxes are initially negatively affected by fire, but quickly recover as vegetation allows use of similar food resources in burned and unburned areas (Cunningham et al. 2006). Mule deer populations elsewhere use all parts of a burn mosaic, including unburned areas for cover and foraging in open areas created by burns (Zimmerman et al. 2006, Horncastle et al. 2013). The smaller

bodied species detected (ground squirrels, chipmunks, woodrats, mice, and gray squirrels) were generalist species, so can use resources in different burn severities. A trapping study of small mammal response to fire near Fawnskin, CA in the San Bernardino Mountains found that California ground squirrel presence was not affected by fire, chipmunks were found across burn severities with more detections in milder burns, and *Peromyscus* detections across burn severity varied by species (Borchert 2014). California ground squirrel, *Peromyscus*, dusky footed wood rats, and chipmunk occupancy in the Sierra Nevada Mountains was minimally affected by fire (Lee and Tietje 2005, Meyer et al. 2007, Roberts et al. 2015). *Peromyscus* in South Dakota benefited from low severity burns (Bock and Bock 1983). Flying squirrels, though more specialized, are likely still able to occupy higher burn severities if key habitat features are intact (Roberts et al. 2015). Four species were only detected at lower burn severities. Bobcats, coyotes, mountain lions, and raccoons all had a limited number of detections and were not attracted to hair tubes, meaning their detection was less likely. These species may occur in different burn severities, but simply were not detected, so different survey methods would be needed to determine this.

Species richness was found to be highest at lower tree mortalities. Nine species were detected across all tree mortalities. Large animals once again can easily travel across areas with varying mortality levels. Small, generalist species can utilize all mortalities. The four species with few detections were once again detected in a limited range but could potentially occur across all severities if surveyed using a different method. Flying squirrels were the one exception, only being detected at lower tree mortalities. Flying squirrels are forest specialists that rely on mature, live trees and are

not able to inhabit areas without them (Smith 2007, Roberts et al. 2015). Similar trends regarding forest age and species richness were found in other forests. Species richness increased in mature mixed wood stands in Alberta compared to younger stands, which lacked food and cover for many species (Roy et al. 1995, Simon et al. 1998).

Conclusions

Both cameras and hair tubes proved useful in detecting mammalian species, Cameras detected a much broader range of species than hair tubes, but detections were less likely without a bait source. Even though hair tubes detected fewer species overall, they proved very useful for detecting those species that were attracted to the bait and were the right size to leave hair samples (Lindenmeyer et al. 1999). Camera traps also proved to have a higher monetary cost compared to hair tubes (Gurnell et al. 2004, Fimbel and Freed 2008), but hair tubes required more effort in processing and identifying samples (Fimbel and Feed 2008, Bertolino et al. 2009). Both methods work very well for detecting species on their own, but using them in tandem greatly increases their effectiveness, with hair tubes acting as an attractant to many species and detecting them in cases when cameras might fail, while cameras were able to detect other species that hair tubes did not. The use of multiple survey methods helps to detects species of different size classes, in different parts of the same area (arboreal vs. terrestrial), allows more opportunities for detection, and avoids detection bias due the method used (Laurance 1992, Li et al. 2012).

Mammalian reactions to and use of areas affected fire vary greatly by species (Doumas and Koprowski 2013, Gwinn and Koprowski 2017). Large bodied mammals

can move in and out of these areas more easily and able to use all the resources available across a mosaic burn (Cunningham et al. 2003, Horncastle et al. 2013). Small bodied generalists are also able to utilize more of these burned landscapes due to their lack of reliance on any specific resource (Bock and Bock 1983, Gwinn and Koprowski 2017). Other more specialized species are still able to exist in a post fire landscape if certain habitat features remain intact (Blount and Koprowski 2012). Maintenance of forest patches after a fire event is key to preserving the species richness in an area (Smith et al. 2000). These forested areas not only provide habitat for forest species, but also act as a source for repopulation (Watson et al. 2012, Robinson et al. 2013). Even if fire severity and regularity increase with climate change the retention of forest refugia can maintain forest ecosystems.

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References

BEKOFF, M. 1977. *Canis latrans*. MAMMALIAN SPECIES 79:1-9.

BERTOLINO, S., L. A. WAUTERS, A. PIZZUL, A. MOLINARI, P. LURZ, AND G. TOSI. 2009. A general approach of using hair-tubes to monitor the European red squirrel: a method applicable at regional and national scales. Mammalian Biology-Zeitschrift für Säugetierkunde 74:210–219.

BLOUNT, S.J. AND J.L. KOPROWSKI. 2012. Response of the Mount Graham red squirrel (*Tamiasciurus hudsonicus grahamensis*) to post fire conditions. Southwestern Naturalist 57:8-5.

BOCK, C.E., AND J.H. BOCK. 1983. Responses of birds and deer mice to prescribed burning in ponderosa pine. Journal of Wildlife Management 47:836–840.

BONNICKSEN, T.M., AND E.C. STONE. 1982. Reconstruction of a presettlement giant sequoia-mixed conifer forest community using the aggregation approach. Ecology 63:1134-1148.

BORCHERT, M.I. 2012. Mammalian carnivore use of a high-severity burn in conifer forests in the San Bernardino Mountains of Southern California, USA. Hystrix 23:50-56.

BORCHERT, M.I. 2014. Responses of small mammals to wildfire in a mixed conifer forest in the San Bernardino Mountains, California. Bulletin of the Southern California Academy of Sciences113:81-95.

BRONGO, L.L., M.S. MITCHELL, AND J.B. GRAND. 2005. Long-term analysis of survival, fertility, and population growth rate of black bears in North Carolina. Journal of Mammalogy 86:1029-1035.

BROWN, J. K., AND J. K. SMITH, EDITORS. 2000. Wildland fire in ecosystems: effects of fire on flora. United States Department of Agriculture Forest Service, Rocky Mountain Research Station, Ogden, Utah, RMRS-GTR-42-vol. 2

CAREY, A. B. 1995. Sciurids in Pacific Northwest managed and old-growth forests. Ecological Applications 5:648-661.

CATTET, M., J. BOULANGER, G. STENHOUSE, R. A. POWELL, AND M. J. REYNOLDS-HOGLAND. 2008. An evaluation of long-term capture effect in ursids: Implications for wildlife welfare and research. Journal of Mammalogy 89:973-990.

CHOATE, D.M., M.L. WOLFE, AND D.C. STONER. 2006. Evaluation of cougar population estimators in Utah. Wildlife Society Bulletin 34:782-799.

CONVERSE, S.J., W.M. BLOCK, AND G.C. WHITE. 2006. Small mammal population and habitat responses to forest thinning and prescribed fire. Forest Ecology and Management 228:263-273.

CUNNINGHAM, S.C., W.B. BALLARD, L.M. MONROE, J.J. RABE, AND K.D. BRISTOW. 2003. Black bear habitat use in burned and unburned areas, central Arizona. Wildlife Society Bulletin 31:786-792.

CUNNINGHAM, S.C., L. KIRKENDALL, AND W. BALLARD. 2006. Gray fox and coyote abundance and diet responses after a wildfire in central Arizona. Western North American Naturalist 66:169-180.

DE MARINIS, A. M., AND A. ASPREA. 2006. Hair identification key of wild and domestic ungulates from southern Europe. Wildlife Biology 12:305–320.

DENNISON, P.E., S.C. BREWER, J.D. ARNOLD, AND M.A. MORITZ. 2014. Large wildfire trend in the western United States, 1984-2011. Geophysical Research Letters 41:2928-2933.

DOUMAS, S.L., AND J.L. KOPROWSKI. 2013. Return of fire as a restoration tool: long-term effects of burn severity on habitat use by Mexican fox squirrels. Restoration Ecology 21:133-139.

FAHRIG, L., AND G. MERRIAM. 1985. Habitat patch connectivity and population survival. Ecology 66:1762-1768.

FAHRIG, L., AND G. MERRIAM. 1994. Conservation of fragmented populations. Conservation Biology 8:50-59.

FIMBEL, C. AND S. FREED. 2008. Monitoring western gray squirrels for landscape management in western Washington. Northwest Science 82:299–308.

FISHER, J. T., AND L. WILKINSON. 2005. The response of mammals to forest fire and timber harvest in the North American boreal forest. Mammal Review 35:51–81.

GURNELL J, P. W. W. LURZ, M. D. F. SHIRLEY, S. CARTMEL, P. J. GARSON, L. MAGRIS, J. STEELE. 2004. Monitoring red squirrel *Sciurus vulgaris* and grey squirrel *Sciurus carolinensis* in Britain. Mammal Review 34:51–74.

GWINN, R. N., AND J. L. KOPROWSKI. 2017. Differential response to fire by an introduced and an endemic species complicates endangered species conservation. Hystrix 27:1-7.

HALOFSKY, J.E., D.C. DONATO, D.E. HIBBS, J.L. CAMPBELL, M.D. CANNON, J.B. FONTAINE, J.R. THOMPSON, R.G. ANTHONY, B.T. BOREMANN, L.J. KAYES, B.E. LAW, D.L. PETERSON, AND T.A. SPIES. 2011. Mixed-severity fire regimes: lessons and hypotheses from the Klamath-Siskiyou Ecoregion. Ecosphere 2:40.

HOBBS, N.T. AND R.A. SPOWART. 1984. Effects of prescribed fire on nutrition of mountain sheep and mule deer during winter and spring. The Journal of Wildlife Management 48:551-560.

HORNCASTLE, V.J., R. F. YARBOROUGH, B.G. DICKSON, AND S.S. ROSENSTOCK. 2013. Summer habitat use by adult female mule deer in a restoration-treated ponderosa pine forest. Wildlife Society Bulletin 37:707-713.

KOPROWSKI, J. L., K. M. LEONARD, C. A. ZUGMEYER, AND J. L. JOLLEY. 2006. Direct effects of fire on endangered Mount Graham red squirrels. Southwestern Naturalist 51:59–63.

LAURANCE, W.F. 1992. Abundance estimates of small mammals in Australian tropical rainforest: a comparison of four trapping methods. Wildlife Research 19:651-655.

LEE, D.E., W.D. TIETJE. 2005. Dusky-footed woodrat demography and prescribed fire in California oak woodland. Journal of Wildlife Management 69:1211-1220.

LEHMKUHL, J. F., K. D. KISTLER, J. S. BEGLEY, AND J. BOULANGER. 2006. Demography of northern flying squirrels informs ecosystem management of western interior forests. Ecological Applications 16:584-600.

LINDENMAYER, D. B., R. D. INCOLL, R. B. CUNNINGHAM, M. L. POPE, C. F. DONNELLY, C. I. MACGREGOR, C. TRIBOLET, AND B. E. TRIGGS. 1999. Comparison of hair tube types for the detection of mammals. Wildlife Research 26:745–753.

LI, S., W.J. MCSHEA, D. WANG, J. HUANG, AND L. SHAO. 2012. A direct comparison of camera-trapping and sign transects for monitoring wildlife in the Wanglang National Nature Reserve, China. Wildlife Society Bulletin 36:538-545.

Lotze, J.H., and S. Anderson. 1979. *Procyon lotor*. Mammalian Species 119:1-8.

MEYER, M.D., D.A. KELT, AND M.P. NORTH. 2007. Effects of burning and thinning on lodgepole chipmunks (*Neotamias speciosus*) in the Sierra Nevada, California. Northwestern Naturalist 88:61-72.

MULCAY, D. M. 2003. Surgical implantation of transmitters into fish. Institute for Laboratory Animal Research Journal 44:295-306.

NAKASHIMA, Y., K. FUKASWA, AND H. SAMEHIMA. 2018 Estimating animal density without individual recognition using information derivable exclusively from camera traps. Journal of Applied Ecology 55:735-744.

OWEN, G., J.D. MCLEOD, C.A. KOLDEN, D.B. FERGUSON, AND T.J. BROWN. 2012. Wildfire management and forecasting fire potential: The roles of climate information and social networks in the Southwest United States. Weather, Climate, and Society 4:90-102

PATTEN, M. A., AND J. C. BURGER. 2018. Reserves as double-edged sword: Avoidance behavior in an urban-adjacent wildland. Biological Conservation 218:233-239.

PICKERELL, G. A., C. F. J. O'DONNELL, D. J. WILSON, AND P. J. SEDDON. 2014. How can we detect introduced mammalian predators in non-forest habitats? A comparison of techniques. New Zealand Journal of Ecology 38: 86–102.

ROBINSON, N.M., S.W.J. LEONARD, E.G. RITCHIE, M. BASSETT, E.K. CHIA, S. BUCKINGHAM, H. GIBB, A.F. BENNETT, AND M.F. CLARKE. 2013. Refuges for fauna in fire-prone landscapes: their ecological function and importance. Journal of Applied Ecology 50:1321-1329.

ROBERTS, S.L., D.A. KELT, J.W. VAN WAGTENDONK, A.K. MILES, AND M.D. MEYER. 2015. Effects of fire on small mammal communities in frequent-fire forests in California. Journal of Mammalogy 96:107-119.

ROY, L., J.D. STELFOX, AND J.W. NOLAN.1995. Relationships between mammal biodiversity and stand age and structure in aspen mixed wood forests in Alberta. In: Relationships Between Stand Age, Stand Structure, and Biodiversity in Aspen Mixed wood Forests in Alberta (Ed. by J.B. Stelfox), pp. 159–189. Jointly Published by Alberta Environment Centre (AECV95-R1), Vegreville, AB and Canadian Forest Service (Project No 00014), Edmonton, AB.

SIMON, N.P., F.E. SCHWAB, E.M. BAGGS, AND G.I. MCTAGGART-COWAN. 1998. Distribution of small mammals among successional and mature forest types in western Labrador. Canadian Field-Naturalist 112:441-445.

SMITH, J.K., L.J. LYON, M.H. HUFF, R.G. HOOPER, E.S. TELFER, AND D.S. SCHREINER. 2000. Wildland fire in ecosystems: effects of fire on fauna. United States

Department of Agriculture Forest Service, General Technical Report RMRS-GTR-42-1:1–84.

SMITH, W. P. 2007. Ecology of *Glaucomys sabrinus*: habitat, demography, and community relations. Journal of Mammalogy 88:862-881.

STACEY, P. B., AND M. TAPER. 1992. Environmental variation and the persistence of small populations. Ecological Applications 2:18-29.

STRATMAN M.R., AND M.R. PELTON. 2007. Spatial response of American black bears to prescribed fire in Northwest Florida. International Association for Bear Research and Management 1:62-71.

TAYLOR, R. J. 1985. Identification of the hair of Tasmanian mammals. Papers and Proceedings of the Royal Society of Tasmania 119:69–82.

TEERINK, B.J. 1991. Hair of West European Mammals: Atlas and Identification Key. Cambridge University Press, Cambridge, UK.

U.S. GLOBAL CHANGE RESEARCH PROGRAM. 2009. Global climate change impacts in the United States. U.S. Global Change Research Program. Thomas R. Karl, Jerry M. Melillo, and Thomas C. Peterson, (eds.). Cambridge University Press, 2009.

WATERS, J. R., AND C. J. ZABEL. 1995. Northern flying squirrel densities in fir forests of northeastern California. Journal of Wildlife Management 59:858-866.

WATSON, S.J., R.S. TAYLOR, D.G. NIMMO, L.T. KELLY, M.F. CLARKE, & A.F. BENNETT. 2012. The influence of unburnt patches and distance from refuges on post-fire bird communities. Animal Conservation, 15:499-507.

WESTERLING, A. L., H. G. HIDALGO, D. R. CAYAN, AND T. W. SWETNAM. 2006. Warming

and earlier spring increase western US forest wildfire activity. Science 313:940–

943.

YOON, J, S. WANG, R. R. GILLIES, L. HIPPS, B. KRAVIT, AND P. J. RASCH. 2015. Extreme

fire season in California: A glimpse into the future. Bulletin of the American

Meteorological Society 96:S1-5.

ZIMMERMAN, T.J., J.A. JENKS, AND D.M. LESLIE JR. 2006. Gastrointestinal morphology of

female white-tailed and mule deer: effects of fire, reproduction, and feeding type.
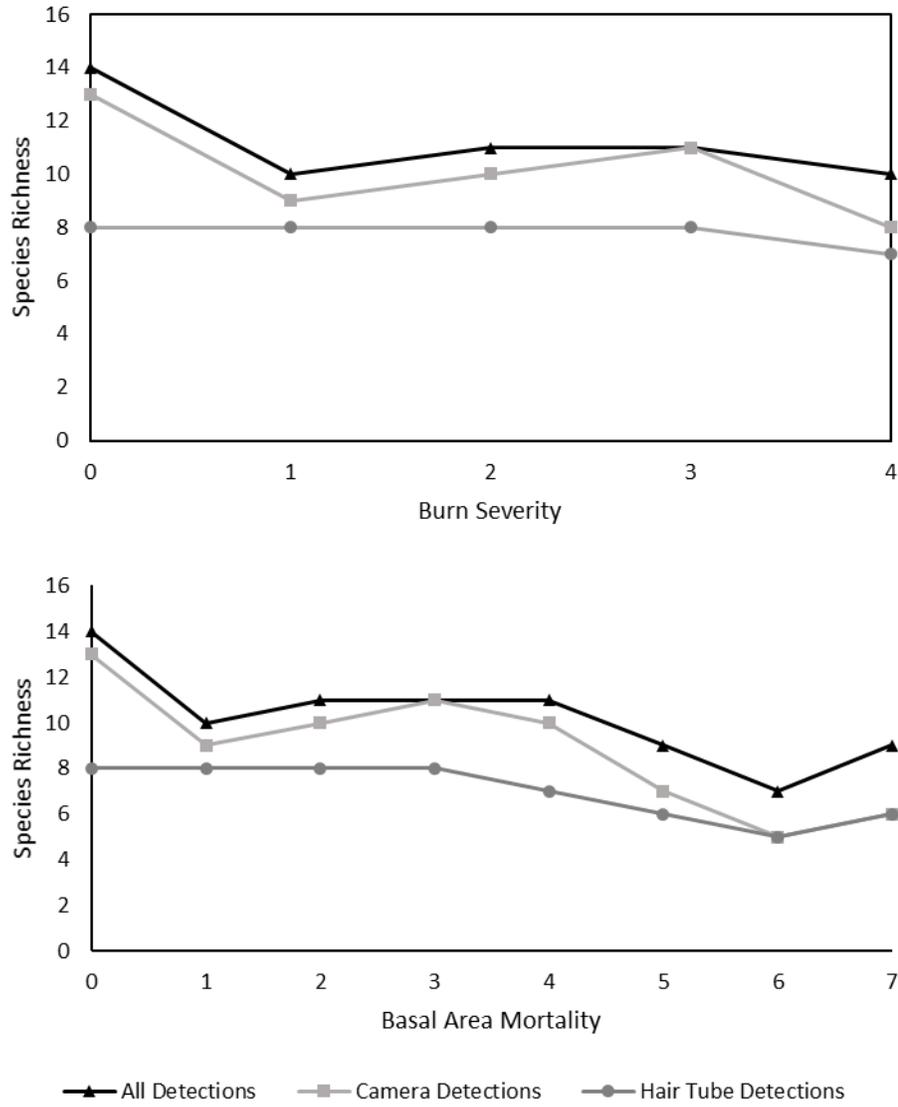
Journal of Mammalogy 87:598-605.

Figure 1-Species Richness for each survey type (camera, hair tube, and all) for burn severity and tree mortality. Burn severity 0 represents the unburned study area and severities 1-4 were in the burned area (1=unburned to low, 2=low burn severity, 3=moderate burn severity, 4=high burn severity). Tree mortality 0 represents the unburned study area and severities 1-7 were in the burned area (1= 0%, 2= 1-10%, 3= 10-25%, 4= 25-50%, 5= 50-75%, 6= 75-90%, 7= >90%).
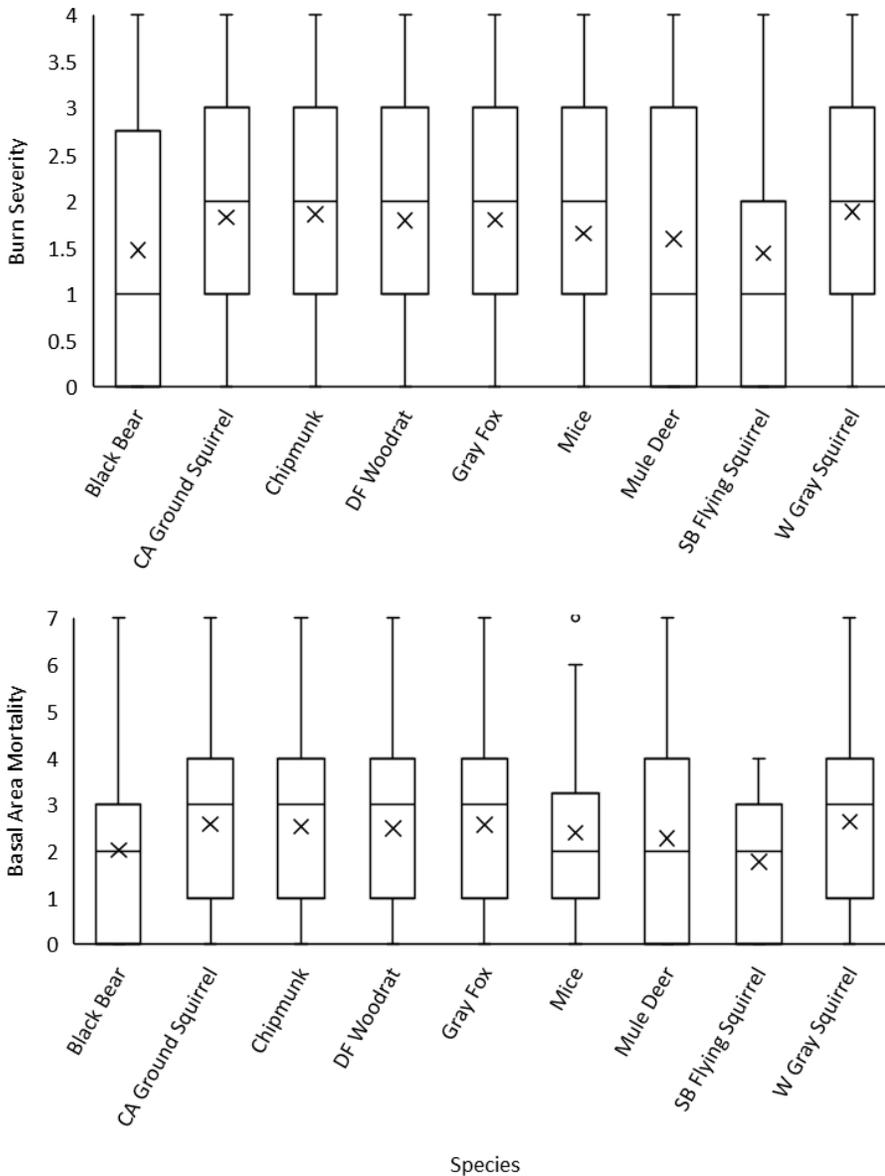
Figure 2- Box and whisker plots showing the minimum, maximum, and mean burn severity and tree mortality for species with >30 detections. Burn severity 0 represents the unburned study area and severities 1-4 were in the burned area (1=unburned to low, 2=low burn severity, 3=moderate burn severity, 4=high burn severity). Tree mortality 0 represents the unburned study area and severities 1-7 were in the burned area (1= 0%, 2= 1-10%, 3= 10-25%, 4= 25-50%, 5= 50-75%, 6= 75-90%, 7= >90%).

Table 1- Total number of surveys with detections (Total), the number of study sites with detections (# Sites), the number of detections by cameras (# Camera), and the number of detections by hair tubes (# Hair Tube) for the 14 species detected.

Species	Detections			
	Total	# Sites	# Camera	# Hair Tube
Black Bear (*Ursus americanus*)	48	36	8	40
Bobcat (*Lynx rufus*)	15	10	15	0
California Ground Squirrel (*Otospermophilus beecheyi*)	604	138	85	588
Chipmunks (*Tamias spp.*)	661	139	265	523
Coyote (*Canis Latrans*)	1	1	1	0
Dusky Footed Woodrat (*Neotoma fuscipes*)	511	130	101	196
Gray Fox (*Urocyon cinereoargenteus*)	66	39	62	4
Mice (*Chaetodipus spp., Reithrodontomys spp., and Peromyscus spp.*)	132	46	132	0
Mountain Lion (*Puma concolor*)	3	3	3	0
Mule Deer (*Odocoileus hemionus*)	78	39	78	0
Raccoon (*Procyon lotor*)	3	2	3	0
San Bernardino Flying Squirrel (*Glaucomys oregonensis californicus*)	357	76	145	286

Skunks (*Mephitis spp.* and *Spilogale spp.*)	28	27	1	27
Western Gray Squirrel (*Sciurus griseus*)	301	121	61	254

Table 2- Minimum (Min), maximum (Max), mean and standard deviation (SD) values of burn severity and tree mortality for the 14 species detected. Burn severity 0 represents the unburned study area and severities 1-4 were in the burned area (1=unburned to low, 2=low burn severity, 3=moderate burn severity, 4=high burn severity). Tree mortality 0 represents the unburned study area and severities 1-7 were in the burned area (1= 0%, 2= 1-10%, 3= 10-25%, 4= 25-50%, 5= 50-75%, 6= 75-90%, 7= >90%).

Species	Burn Severity				Tree mortality			
	Min	Max	Mean	SD	Min	Max	Mean	SD
Black Bear (*Ursus americanus*)	0	4	1.47	1.19	0	7	2.03	1.82
Bobcat (*Lynx rufus*)	0	3	1.20	1.47	0	4	1.30	1.62
California Ground Squirrel (*Otospermophilus beecheyi*)	0	4	1.83	1.27	0	7	2.59	1.96
Chipmunks (*Tamias spp.*)	0	4	1.86	1.30	0	7	2.53	1.95
Coyote (*Canis latrans*)	0	0	0.00	0.00	0	0	0.00	0.00
Dusky Footed Woodrat (*Neotoma fuscipes*)	0	4	1.79	1.29	0	7	2.48	1.95
Gray Fox (*Urocyon cinereoargenteus*)	0	4	1.79	1.24	0	7	2.56	1.96
Mice (*Chaetodipus spp., Reithrodontomys spp., and Peromyscus spp.*)	0	4	1.65	1.20	0	7	2.39	1.92
Mountain Lion (*Puma concolor*)	0	2	0.67	.094	0	2	0.67	0.94
Mule Deer (*Odocoileus hemionus*)	0	4	1.59	1.30	0	7	2.28	1.95
Raccoon (*Procyon lotor*)	0	0	0.00	0.00	0	0	0	0.00

San Bernardino Flying Squirrel (*Glaucomys oregonensis californicus*)	0	4	1.47	1.20	0	4	1.78	1.40
Skunks (*Mephitis spp.* and *Spilogale spp.*)	0	4	1.37	1.31	0	7	1.93	1.84
Western Gray Squirrel (*Sciurus griseus*)	0	4	1.88	1.31	0	7	2.64	1.98