

Acoustic Inventory and Monitoring of Bat Species In the Power Fire Burn Area – 2014, 2015, & 2016 Field Seasons

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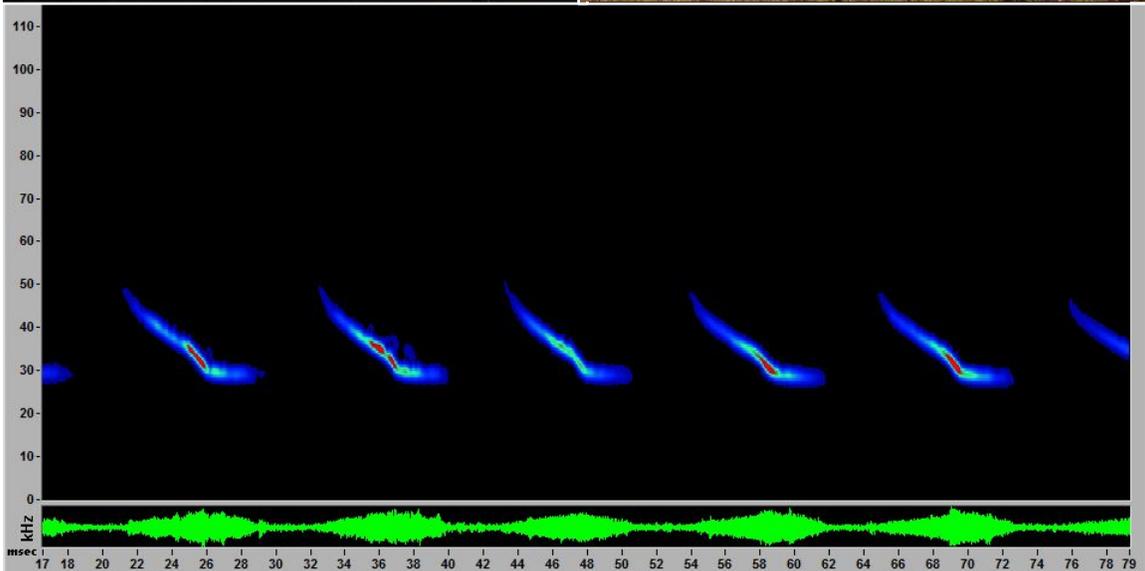


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Executive Summary

During 2014-2016 bat monitoring was conducted in and around the 2004 Power Fire area using automated recording units. This report summarizes the data collected during those monitoring efforts, and presents analysis of the influence of the fire and post-fire management on the forest bat community. In total, 47 unique locations (accounting for 526 survey-nights) were surveyed, including sites outside of the burn perimeter, unchanged/low severity sites, moderate severity sites, and high severity sites that had either been managed for reforestation or left unmanaged. Surveys were conducted over multiple nights during each deployment of recording units, and most sites were surveyed during multiple years. An analysis of data collected from these efforts produced the following primary results:

- All 17 Sierra Nevada bat species were observed at least once either within the Power Fire or at nearby unburned sites. The most frequently observed species were the Mexican free-tailed bat (*Tadarida brasiliensis*), and California myotis (*Myotis californicus*). Of the three Eldorado National Forest USFS sensitive species, fringed myotis (*M. thysanodes*) and pallid bat (*Antrozous pallidus*) were observed during each of the three monitoring seasons in both burned and unburned locations. Townsend's big-eared bat (*Corynorhinus townsendii*) was observed only once at an unburned location.
- The amount of bat activity recorded during any given night ranged widely (95% interval = 1 to 302 passes/night) with a nightly mean and median of 95 and 52 bat passes, respectively. Mean nightly bat activity was greater during the 2016 season (109 passes/night) than 2015 (88 passes/night). During 2014 we observed 70 bat passes per night on average, but surveys were conducted over a smaller part of the season during this year.
- A comparison of burned and unburned locations showed higher bat species diversity per survey within the burned area. Activity levels of individual species were also significantly higher within the burn for silver-haired bat (*Lasionycteris noctivagans*) and fringed myotis.
- Average species richness did not vary across the range of burn severity sampled. However, the composition of the local bat community shifted somewhat as burn severity increased from low to high severity. In total, six species showed significant relationships with burn severity. Mexican free-tailed bat, silver-haired bat, and hoary bat (*Lasiurus cinereus*) activity increased with burn severity, while activity of long-eared myotis (*M. evotis*), small-footed myotis (*M. ciliolabrum*), and pallid bat decreased.
- Post-fire management of high severity areas in the form of salvage logging and subsequent planting of desirable tree species did not appear to affect mean species richness. However, there appears to be greater variation in bat diversity among unmanaged than managed high severity sites. At the species level, activity of hoary bat was greater in unmanaged areas, but differences for other species were not statistically significant.

In general, the bat community of Eldorado National forest appears to have benefited from the Power Fire and the habitat heterogeneity it created. Wildfire that creates variable stand densities, and produces some large snags, likely improves both foraging and roosting habitat for a number of species. Bats of the Sierra Nevada are likely best adapted to wildfires that fall within the natural range of variation in terms of severity levels and patch sizes. Management that maintains and promotes habitat heterogeneity and the long-term availability of roosting sites will best benefit the bat community within and around the Power Fire.

Introduction

Bats are a diverse and functionally important part of the wildlife community in Sierra Nevada forests. Of the 23 bat species found in California, 17 utilize Sierra Nevada habitats. Among these species, 13 are known to use trees for roosting at some point in their lifecycle (Brigham 2007); habitat features that are directly affected by wildfire, and forest management. The species found in the Sierra Nevada feed nearly exclusively on insects, and consume large amounts of insect biomass every night (Brigham 2007). In this way bats provide an ecosystem service in the form of pest-reduction, which has been shown to have significant economic value in agricultural systems (Boyles et al. 2011). Bats may suppress pest outbreaks in forest systems as well, but the magnitude of this effect has yet to be evaluated. Similarly, it has been proposed (but not yet tested) that bats are “fertilizers of the forest” bringing nutrients from other ecosystems to forested areas while foraging (Brigham 2007). Bats face a number of threats including destruction of habitat, altered disturbance regimes, climate change, and the spread of white-nosed syndrome across the United States (Lacki et al. 2007b, Frick et al. 2016).

Despite their prominent place among the diversity and ecology of western forests, and the growing threats to bat populations nation-wide, the community is relatively under-studied (Miller et al. 2003). The neglect of bats as a class of wildlife is in part due to the difficulty of monitoring bat occurrence and habitat use. Surveys of roosting and breeding colonies (e.g., caves and bridges), as well as mist net captures of individuals, have been the primary methods of monitoring bat species for many years (Lacki et al. 2007a). Such methods provide valuable and detailed information about the health and reproduction of a select number of species, but the ability to monitor across the range of habitats used by bats is limited. Furthermore, colony surveys and mist-netting can be costly in terms of surveyor effort. In recent years, as technology has improved, the use of automated recording units (ARUs) have grown in prominence as a complimentary survey technique (Frick 2013). ARUs record the echolocation calls of foraging and commuting bats, which can be later classified to species using associated software. ARUs can be deployed for weeks at a time and conduct nightly surveys with only periodic effort by field technicians.

Among the areas where our understanding of bat ecology is particularly limited is their response to fire-altered habitats and forest management (but see Buchalski et al. 2013). Wildfire in the Sierra Nevada is a natural and common disturbance that can dramatically reshape landscapes and the habitats therein (Sugihara et al. 2006). Within individual fires, the effects of the burn on vegetation structure, and ecosystem function can range wildly (Fites-Kaufman et al. 2006). Understanding how this range of fire effects influences bat occurrence and activity is essential for predicting how wildfires such as the 2004 Power Fire affect species of concern. This is especially important given the observed increase in fire size and severity within the Sierra Nevada due to past fire suppression and ongoing climate change (Miller and Safford 2012; Steel et al. 2015; Westerling et al. 2006).

Large burns such as the Power Fire create areas of high priority for management due to concerns regarding maintaining forest cover, mitigating future fire risk, and maintaining

habitat for wildlife communities. Where reforestation efforts are implemented, standing dead trees are often removed within one to two years post-fire, followed by planting of desired tree species (typically pines). Studies assessing the influence of these management actions on the wildlife community have shown mixed effects, with some species responding negatively while for others postfire management effects are more equivocal (Kotliar et al. 2002). One notable finding among these studies is that lowering basal area of fire-killed snags reduces the occurrence of some fire specialist species such as the black-backed woodpecker in the years immediately following the fire (*Picoides arcticus*; Tingley et al. 2014, White et al. 2015). Such an effect may be mirrored by cavity roosting bat species, but to our knowledge, no studies have assessed the impact of salvage logging or reforestation following wildfire on the bat community of the Sierra Nevada. Large burns such as the Power Fire are becoming more common and monitoring their effects on the bat and other wildlife communities will help inform post-fire management, and allow us to better anticipate the effects of future fires in the Eldorado National Forest and the Sierra Nevada as a whole.

Study Site

The study area includes the Power Fire and surrounding unburned mixed conifer forests, located on the Amador Ranger District of the Eldorado National Forest in the Sierra Nevada Mountains of California (Figure 1). The Power Fire burned during the fall of 2004 in the Mokelumne River canyon, south of Highway 88 in Amador County. The fire began on October 6th of that year and was contained on October 16th. A total of 7000 hectares (17,200 acres) burned with heterogeneous effects to existing vegetation. Within the burn perimeter, an assessment of vegetation burn severity classified 7% of the area as unchanged, 22% as low severity, 33% as moderate severity, and 38% as high severity (Figure 1). The Power Fire resulted in a decrease in the area of the mixed conifer vegetation type, and an increase in montane hardwoods and annual/perennial grasslands within its perimeter (Estes and Gross 2015). The burn area ranges from 940 m (3070 ft) to 2100 m (6890 ft) in elevation and is located predominantly on south-facing slopes. The fire burned mostly through the mid-montane zone dominated by white fir (*Abies concolor*), and Ponderosa pine (*Pinus ponderosa*), with incense cedar (*Calocedrus decurrens*), sugar pine (*P. lambertiana*), red fir (*A. magnifica*), and Douglas fir (*Pseudotsuga menziesii*) also occurring (Richter and Safford 2016). Our unburned reference sites were located within the same elevational range and pre-fire forest type.

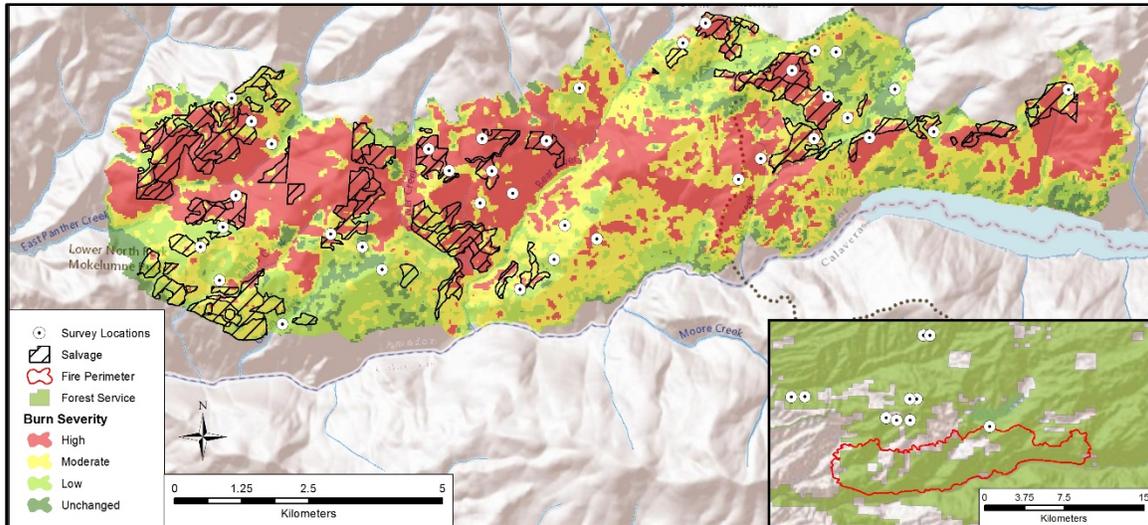


Figure 1. The study area and bat monitoring locations within and near the Power fire. Burn severity levels and areas that were salvage logged are shown within the fire perimeter. The inset map shows unburned reference locations outside of the fire perimeter. Burn severity levels follow Miller and Thode (2007), with the high severity class representing > 95% tree mortality.

Methods

Sample design

Bat survey locations were selected using a stratified random sampling protocol where the population of potential survey locations was limited to 198 avian point count stations within and around the Power Fire perimeter. The avian survey points were part of monitoring projects conducted in parallel with bat monitoring by Point Blue Conservation Science and University of California-Davis scientists. All potential points were located in mixed conifer forests or formerly mixed conifer forests prior to the burn. Co-locating survey efforts allowed us to make efficient use of vegetation data and researcher time. Detailed methods on bird sampling design can be found in Fogg et al. (2015) and Roberts et al. (2011). We grouped the potential sampling locations into five strata for selection: 1) unburned reference points outside the Power Fire perimeter, 2) unchanged/low severity points, 3) moderate severity points, 4) high severity points that had not been managed for reforestation following the fire, and 5) high severity points that had been salvage logged and replanted. All potential unburned reference points were located within the same elevation band as the potential burned survey locations (1230-2016 m; 4034-6613 ft). Within each strata, all points were assigned a random prioritization. We selected points for bat surveys using this prioritization order unless a point was within 500 m of a higher priority site in which case the subsequent point on the list was used. Points were spaced at least 500 m apart to minimize effects of spatial autocorrelation. This sampling protocol aimed to balance sampling effort across the five

focal strata with one site from each group being surveyed each deployment cycle. Ultimately effort was nearly, but imperfectly, balanced due to rare equipment failures or other logistical constraints (Table 1).

Table 1. The number of unique survey locations and total survey nights stratified by severity levels and management. Reforestation points experienced both salvage logging and subsequent replanting.

Sampling Category	Locations	Survey Nights
Reference	10	128
Low Severity	8	95
Moderate Severity ¹	12	115
High Severity – No management	9	97
High Severity – Reforestation	8	91
Total	47	526

¹One moderate severity location (PW08.8) accounting for 7 survey nights was managed for reforestation and was included in analyses of management effects, which are otherwise limited to high severity points.

Burn severity categories were assigned using the USFS Region 5 burn severity database (available at <http://www.fs.usda.gov/wps/portal/fsinternet/main/r5/landmanagement/gis>) based on Miller and Thode (2007). For analysis of burn severity effects we used the continuous Relative delta Normalized Burn Ratio (RdNBR) values on which the ordinal burn severity categories are based. Among the high severity points, management status (i.e. whether the point had been salvage logged and reforested) was assigned using the Forest Service Activity Tracking System (FACTS) database (available at <http://www.fs.usda.gov/detail/r5/landmanagement/gis>), and verified during vegetation surveys (Fogg et al. 2016). All points classified as undergoing active reforestation were both salvage logged and replanted with predominantly pine species following the Power Fire.

Deployment protocol

Bat surveys were conducted using automated recording units (ARUs; SM3BAT model - Wildlife Acoustics inc.) coupled with ultrasonic microphones (SMM-U1 model - Wildlife Acoustics inc.). ARUs were deployed for approximately two weeks at a time (equal to the estimated life of internal batteries), recording bats during alternate nights¹ from 30 minutes prior to sunset to 30 minutes following sunrise. Due to rare equipment failures, and variable battery life, the number of successful survey nights per deployment ranged from 1 to 9 nights with a mean of 5.4 nights. Battery life depends on a number of factors including the amount of bat activity and false triggers, as well as the type of batteries used. Most deployments used internal D batteries, but during each deployment cycle one unit was coupled with a higher capacity external 12-volt battery that allowed for additional survey nights. In 2014 we surveyed 12 locations between July and

¹ Owl calls were recorded using a non-ultrasonic microphone during alternate nights (results not reported here).

September. During 2015 and 2016, deployments were made between May and September with a total of 42 and 43 locations surveyed in respective years. Nine points were visited during one of the three years, twenty-six locations were visited two of the three years, and twelve locations were visited during all three years.

ARUs were secured to vegetation at a survey location (usually a tree or snag) using a chain and combination lock. Ultrasonic microphones were attached to the ARU via a cable and elevated approximately 3m above the ground using a pole supported by vegetation or a piece of rebar. Microphones were located away from branches, other sources of clutter, and sound-reflective surfaces that might distort acoustic recordings. Microphones were pointed toward open areas (relative to the habitat being sampled) to maximize the likelihood of detecting a passing bat (Figure 2). A detailed deployment protocol can be found in Appendix C.



Figure 2. Example automated recording unit and microphone setup.

Processing protocol

Each recorded bat pass was classified using Sonobat version 3.2.1 with the US west license (Szewczak 2010). The software classifies recordings to species when possible and to broader taxonomic groups (e.g. high vs. low frequency species) when recording quality is poor or when discrimination between similar species is inconclusive. Classifications are made by comparing call characteristics of recorded bat passes against a library of known bat calls for 17 California bat species (Table 2; Figure 3). Sonobat also assigns a likelihood of presence for each species during a given night and provides a nightly corrected count of each species. Corrected counts are a conservative estimate of the number of recordings for each species as many low quality or ambiguous recordings are discarded. We use both total unclassified bat passes, and species-level corrected counts in our analysis for this report. Reported species richness values are calculated using corrected counts of individual species. Thus, reported species richness and activity levels of individual species should be considered conservative estimates. Regardless of this cautious approach, some misclassifications during the automated process are still possible at low rates. To estimate the rate of classification error at the survey level we manually vetted a subset of the bat surveys using Sonobat reference calls and documentation (Szewczak 2010). To improve our estimate of classification error rates we also included vetted surveys from parallel monitoring projects in the Rim, Chips, and Storrie fires (*unpublished data*). In total, 101 survey nights across the four projects underwent manual vetting.

Table 2. The common name, scientific name and species code of potential species, ordered alphabetically by common name. Whether each species was observed at least once during each survey year is also noted.

Common Name	Scientific Name	Species Code	Observed		
			2014	2015	2016
Big brown bat	<i>Eptesicus fuscus</i>	epfu	X	X	X
California myotis	<i>Myotis californicus</i>	myca	X	X	X
Fringed myotis*	<i>Myotis thysanodes</i>	myth	X	X	X
Hoary bat	<i>Lasiurus cinereus</i>	laci	X	X	X
Little brown bat	<i>Myotis lucifugus</i>	mylu	X	X	X
Long-eared myotis	<i>Myotis evotis</i>	myev	X	X	X
Long-legged myotis	<i>Myotis volans</i>	myvo	X	X	X
Mexican free-tailed bat	<i>Tadarida brasiliensis</i>	tabr	X	X	X
Pallid bat*	<i>Antrozous pallidus</i>	anpa	X	X	X
Silver-haired bat	<i>Lasionycteris noctivagans</i>	lano	X	X	X
Small-footed myotis	<i>Myotis ciliolabrum</i>	myci		X	X
Spotted bat	<i>Euderma maculatum</i>	euma			X
Townsend's big-eared bat*	<i>Corynorhinus townsendii</i>	coto			X
Western mastiff bat	<i>Eumops perotis</i>	eupe	X	X	X
Western pipistrelle	<i>Parastrellus hesperus</i>	pahe	X	X	X
Western red bat	<i>Lasiurus blossevillii</i>	labl		X	
Yuma myotis	<i>Myotis yumanensis</i>	myyu	X	X	X

* Eldorado National Forest sensitive species.

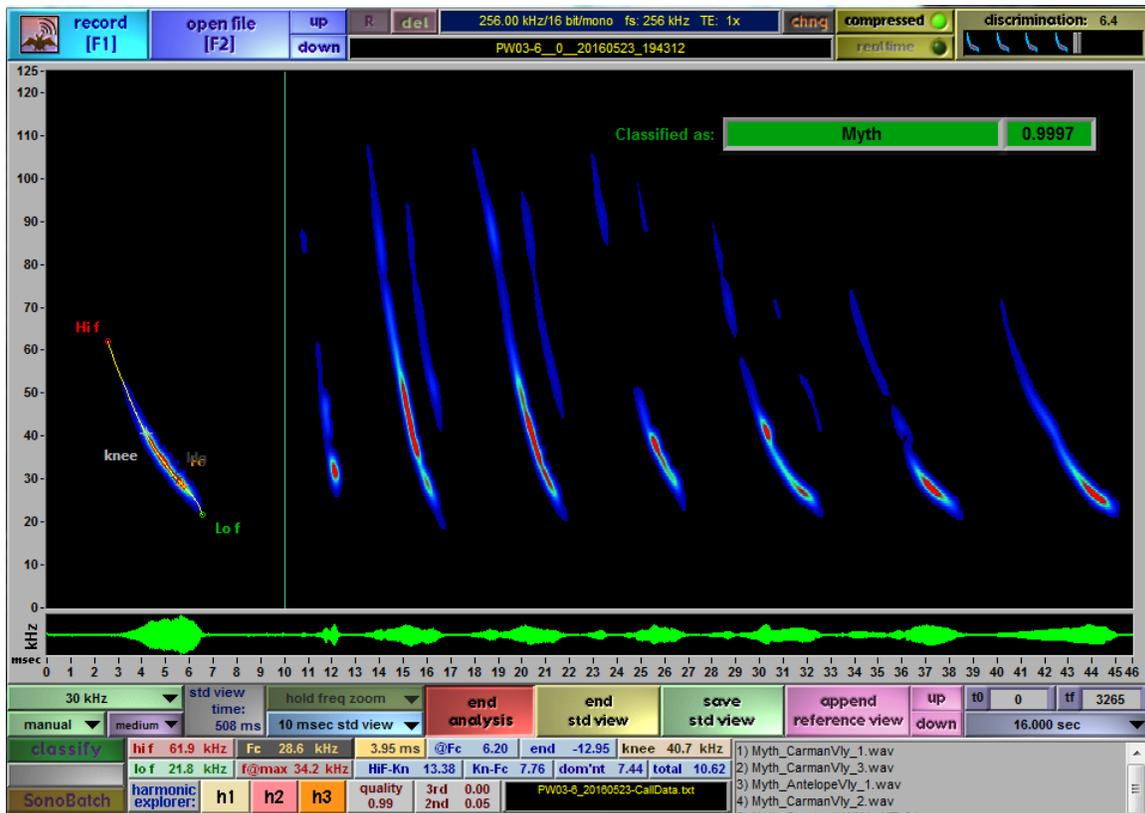


Figure 3. Example sonogram from SonoBat software of a bat call classified as a fringed myotis (*Myotis thysanodes*; left) compared to a range of known call types by the same species (right).

Data Analysis

We assessed the condition of the bat community in and around the Power Fire using the metrics of species richness and activity level. Richness provides a measure of the diversity of the bat community within different habitat types or conditions, while activity level is an indicator of relative use of the habitat by the full bat community or individual species. Activity levels are likely correlated with bat abundance, but are not synonymous with abundance since individual bats may be detected multiple times during a survey. Multiple recordings of the same individual are especially likely if they are continuously using the local area rather than passing through.

In addition to species inventory results, bat community seasonality, and annual variation, we built three families of statistical models, which assessed differences in bat richness and individual species activity levels: 1) between burned and unburned areas, 2) across the gradient of burn severity within the fire perimeter, and 3) between managed (salvaged and replanted) and unmanaged high severity areas. For each statistical model we used generalized linear models with a Poisson error structure and logistic link. Differences and effects estimates are considered significant when 95% confidence intervals do not encompass zero. For all models, point ID and year are included as random intercepts to

avoid issues associated with pseudo-replication due to repeated measures. All analyses were conducted using program R (R Core Team 2017), with the lme4 (Bates et al. 2015) and statistical rethinking (McElreath 2015) packages as our primary tools for model construction.

Results and Discussion

Species inventory & activity levels

We conducted surveys for 17 bat species known to occur in the Sierra Nevada region. During the 2014-2016 surveys, all 17 species were recorded at least once either within the Power Fire perimeter or the nearby unburned forest. Most species were observed during all three survey years with four exceptions: spotted bat (*Euderma maculatum*) and Townsend's big-eared bat (*Corynorhinus townsendii*) were only observed during the 2016 season, and western red bat (*Lasiurus blossevillii*) was only observed during the 2015 season. Small-footed myotis (*Myotis ciliolabrum*) was observed during 2015 and 2016, but not during the abbreviated monitoring period of 2014 (Table 2). The number of species observed during any given survey night ranged from zero to 11 with a mean of 3.5 species per night. Multiple nights of monitoring are conducted during each deployment in part to account for nightly variability in bat activity due to weather, the inconsistent use of specific feeding grounds, variable recording quality due to ambient noise etc. Thus, the aggregations of all repeated surveys likely provide the fullest view of the bat community at a given point. The maximum number of species observed at each point ranged from two to 11 with a mean of 5.7 (Figure 4a). Aggregated corrected counts of each species and each survey location can be found in Appendix A.

The Eldorado National Forest hosts three Forest Service sensitive species²: the fringed myotis (*M. thysanodes*), pallid bat (*Antrozous pallidus*), and Townsend's big-eared bat. Fringed myotis and pallid bat were recorded at 30 (64%) and 17 (36%) survey locations, respectively. Townsend's big-eared bat was only recorded once, outside of the fire perimeter (Figure 5). The species observed across the highest number of points in the study region were the Mexican free-tailed bat (*Tadarida brasiliensis*) and the California myotis (*M. californicus*), which were recorded at all but one and two points, respectively (Appendices A & B).

² http://www.fs.usda.gov/Internet/FSE_DOCUMENTS/stelprdb5435266.xlsx

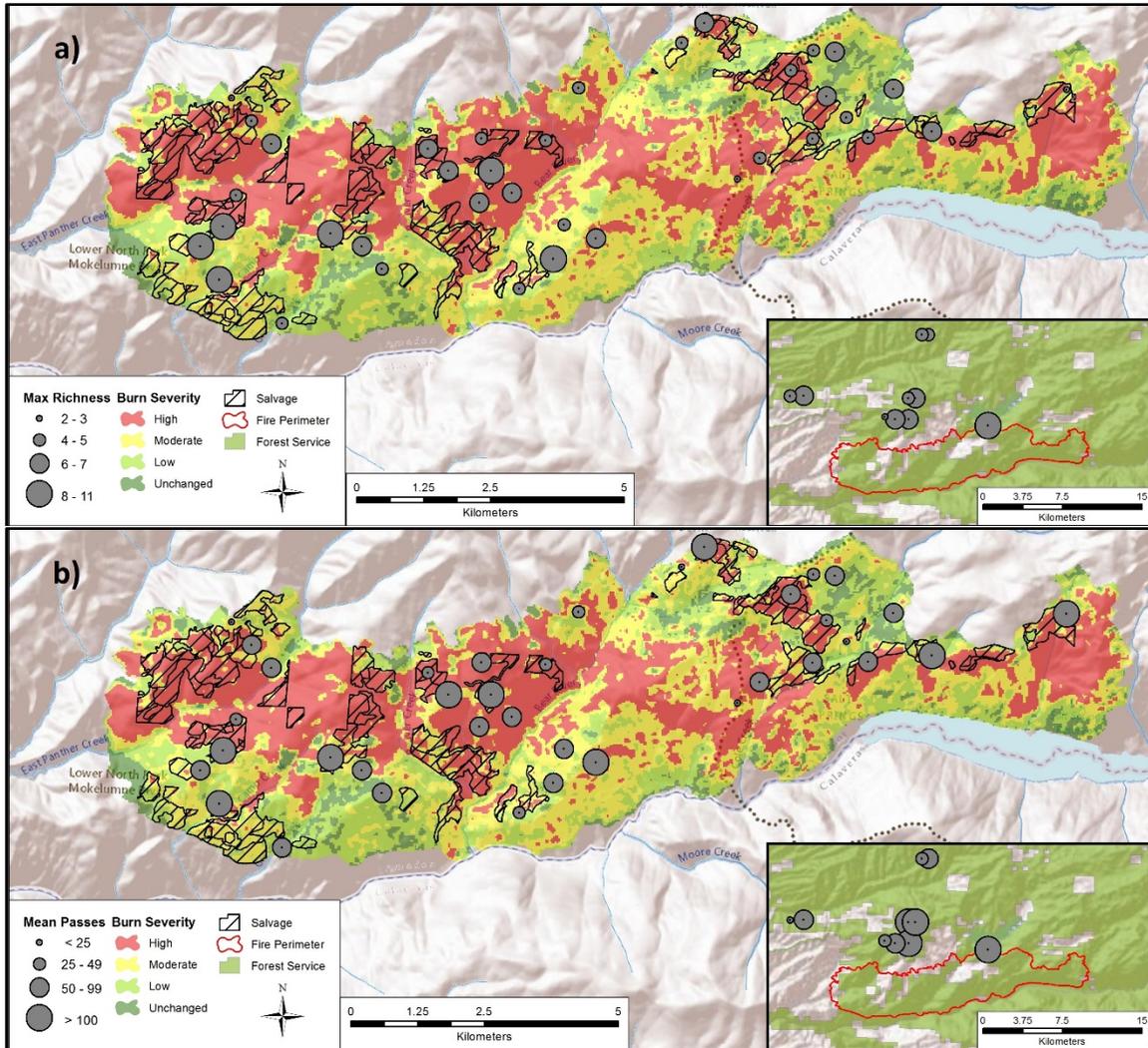


Figure 4. a) Maps of species richness and b) community activity level at each survey location. The size of the circle is relative to the maximum number of species observed across surveys of each location, and the average number of passes observed per survey night, respectively.

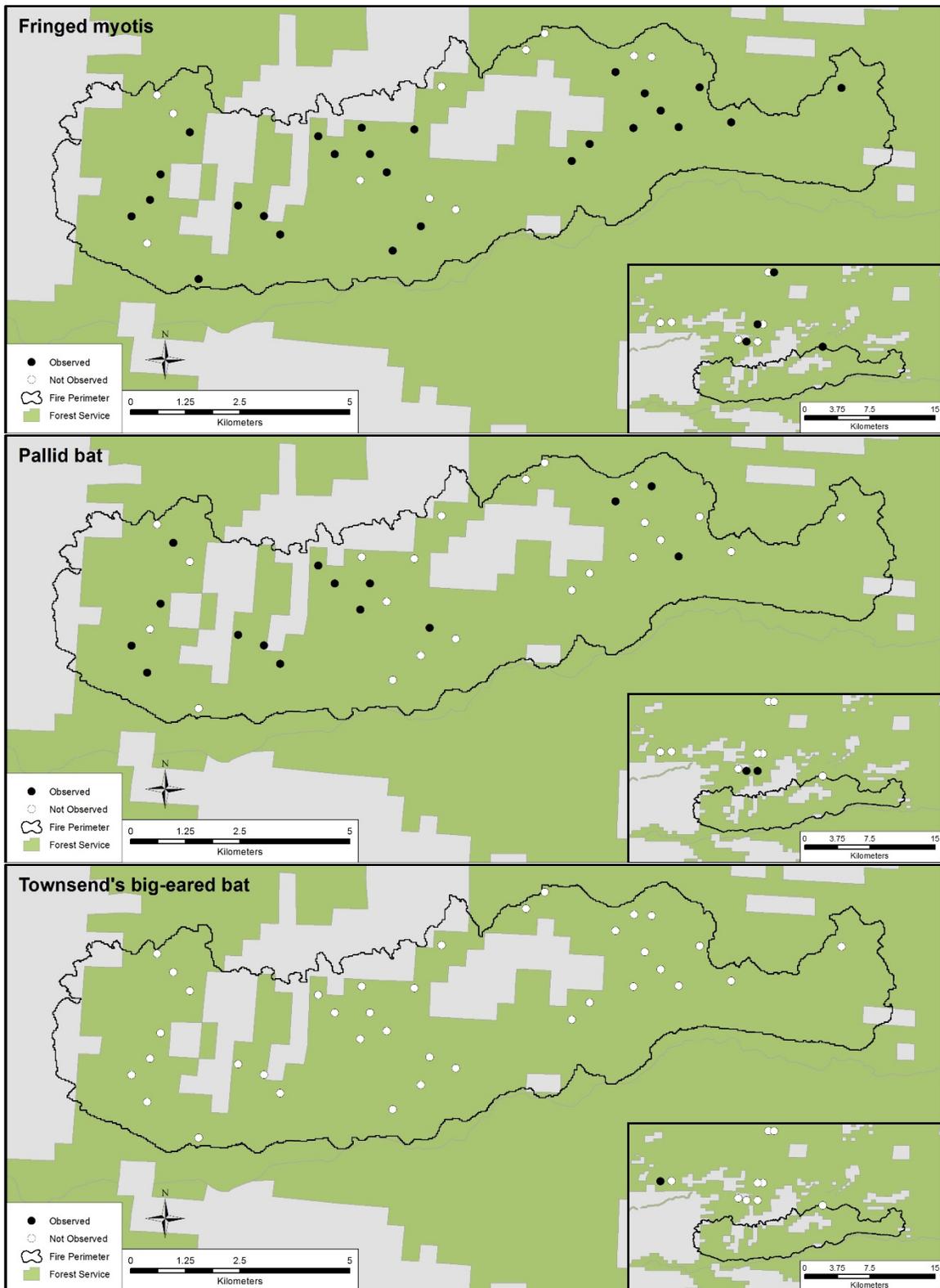


Figure 5. Survey locations where Eldorado National Forest sensitive species were observed at least once during 2014-2016 surveys. Maps for all species can be found in Appendix B.

The amount of bat activity recorded during any given night ranged widely (95% interval = 1 to 302 passes/night) with a nightly mean and median of 95 and 52 bat passes, respectively. The distribution of passes per night appears to follow a log-normal or Poisson distribution where most surveys record relatively few bat passes, but a few record much higher activity levels. The maximum number of passes recorded during a single night was 1667. We recorded significantly more bat activity on average during 2016 (109 passes/night) than 2015 (88 passes/night). 70 passes/night were observed on average during 2014, but this is not included in the inter-annual comparison because surveys were only conducted from late July through September of that year. Within each season, the number of passes varied greatly over time (Figure 6a). With some variation between species, Sierra Nevada bats will generally arrive at their summer grounds during the spring, females will give birth during June and July, juveniles will become volant (able to fly) approximately a month later, and individuals will migrate to winter habitats and/or hibernate beginning in the fall (Richardson 2011). This general pattern of bat seasonality is reflected in our data with peak activity levels occurring in July and the beginning of August when juveniles typically join the population of active foragers. Between 2015 and 2016, activity levels were very similar prior to July 1st, but diverged thereafter (Figure 5a). This departure may indicate superior rearing success of one or more species during 2016.

Activity levels for individual bat species was also highly variable, with Mexican free-tailed bats registering the most corrected counts per night on average (20.5) followed by California myotis (9.3) and silver-haired bat (*Lasionycteris noctivagans*; 5.8). Most species were recorded fewer than once per survey on average (Figure 6b). Townsend's big-eared bat and spotted bat had the lowest apparent activity levels in our surveys with only one corrected count apiece (Appendix A).

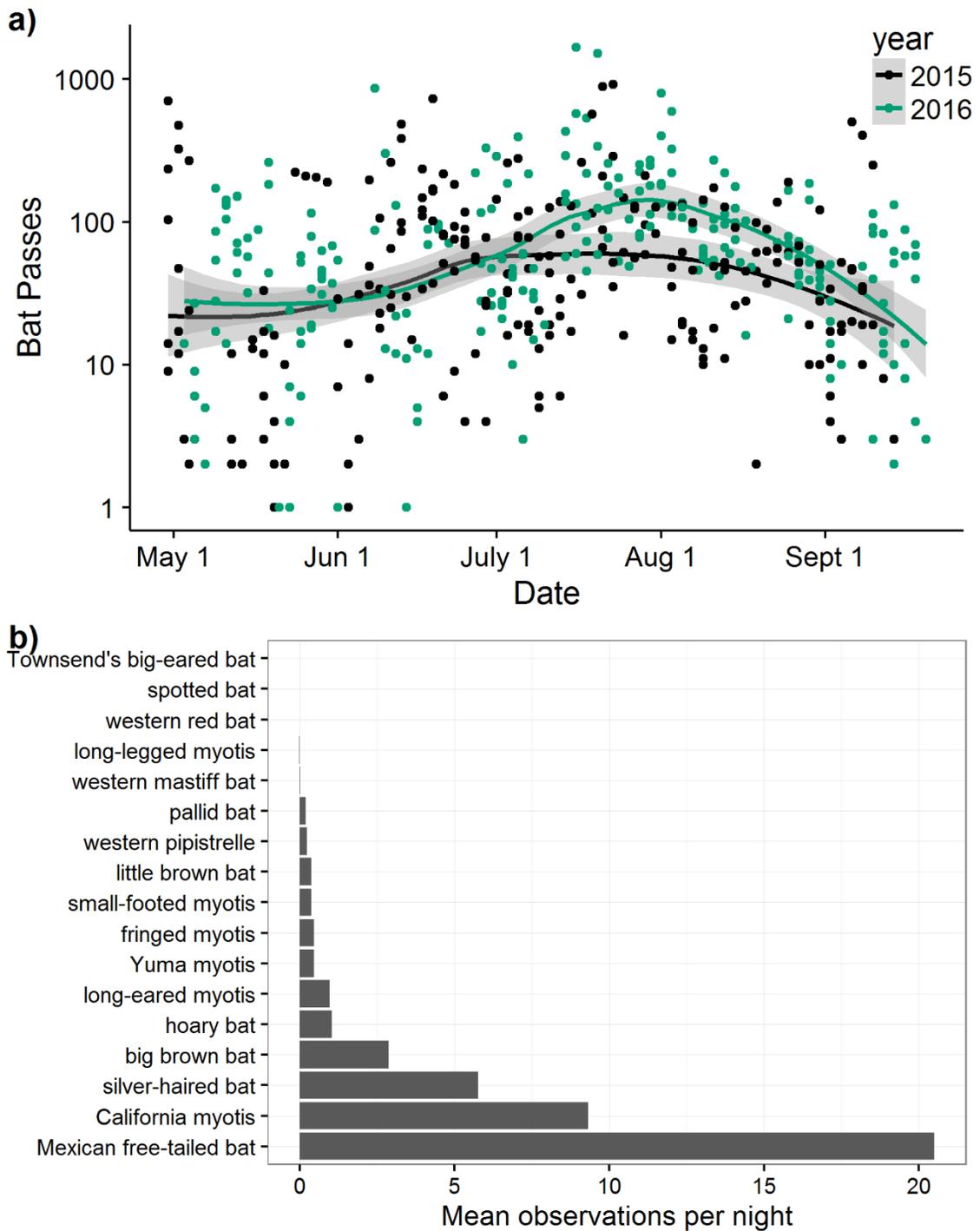


Figure 6. a) Unclassified bat activity (log scale) over time for the of 2015 and 2016 field seasons, with overlaid loess lines and 95% confidence intervals. 2014 surveys occurred during late July through September only and are not displayed. b) Average classified recordings (corrected counts) per survey night across all survey locations and seasons for each species.

Burned vs. Unburned areas

We compared species richness and individual species activity levels between sites within the Power fire (37 locations and 398 survey nights), and those outside of the burn perimeter (10 locations and 128 survey nights; Table 1). Significantly more species were observed during a typical night at points within the fire (median = 3.57) than at points in the unburned forest (median = 2.80; Figure 7a). Many individual species were also observed more often within the burned area than outside of it, with these differences being significant for the silver-haired bat and fringed myotis (Figure 7b & c; Appendix A). Among the most active bat species, the Mexican free-tailed bat appears to favor burned areas, while California myotis was recorded at similar levels in both burned and unburned points (Figure 7b). Among the sensitive species, pallid bat shows a similar (but non-significant) tendency as fringed myotis toward burned areas. The sole Townsend's big-eared bat classification was outside of the fire perimeter (Figures 5 & 7c).

Mixed severity wildfires such as the Power Fire create a heterogeneous post-fire landscape with variable vegetative structure and species compositions (Perry et al. 2011; Richter and Safford 2016). As compared to long-unburned forests with dense and relatively homogenous habitats, it is perhaps unsurprising that more species were recorded on average within the Power Fire and the wide range of habitat types it created. One study of bat activity in and around the 2002 McNally Fire in the southern Sierra Nevada also found that at least some groups of bat species preferred burned to unburned areas (Buchalski et al. 2013). Buchalski et al. suggest the observed differences may be attributable to increased availability of prey and roosting sites within burned areas as well as reduced clutter (i.e., more open flyways), which facilitates foraging for large-bodied bat species.

Burn severity effects

While differences in species richness were apparent between burned and unburned locations, bat diversity showed no directional change across the range of burn severity sampled within the Power Fire (Figure 8a). Our model of burn severity effects predicts mean nightly observations of slightly less than 4 species on average regardless of burn severity. However, individual species models provide evidence that the composition of the bat community changes with burn severity. The effect of severity was significant for six species in total. The activity levels of Mexican free-tailed bat, silver-haired bat, and hoary bat appear to increase with higher levels of burn severity, while the opposite is true for long-eared myotis (*M. evotis*), small-footed myotis, and pallid bat (Figure 8b & c; Appendix A). The USFS sensitive species fringed myotis was found more often in burned areas than unburned areas (Figure 7b), but its activity level did not vary significantly across the range of burn severity within the Power Fire (Appendix A). 37 unique points accounting for 398 survey nights (Table 1) were used in our assessment of burn severity effects.

Observed differences between species across the gradient of burn severity may be in part attributable to foraging strategies and morphologies of the species observed. In general, small-bodied bat species with low wing-loading (body mass / wing area) such as the long-eared myotis and small-footed myotis, are more maneuverable and can effectively

hunt in cluttered environments. This is consistent with their apparent affinity to the relatively dense forests found in low severity areas of the Power Fire. In contrast, large-bodied species with high wing-loading such as the Mexican free-tailed bat and hoary bat are less agile species, and are known to prefer more open habitats (Lacki et al. 2007b) such as the high severity areas of the Power Fire.

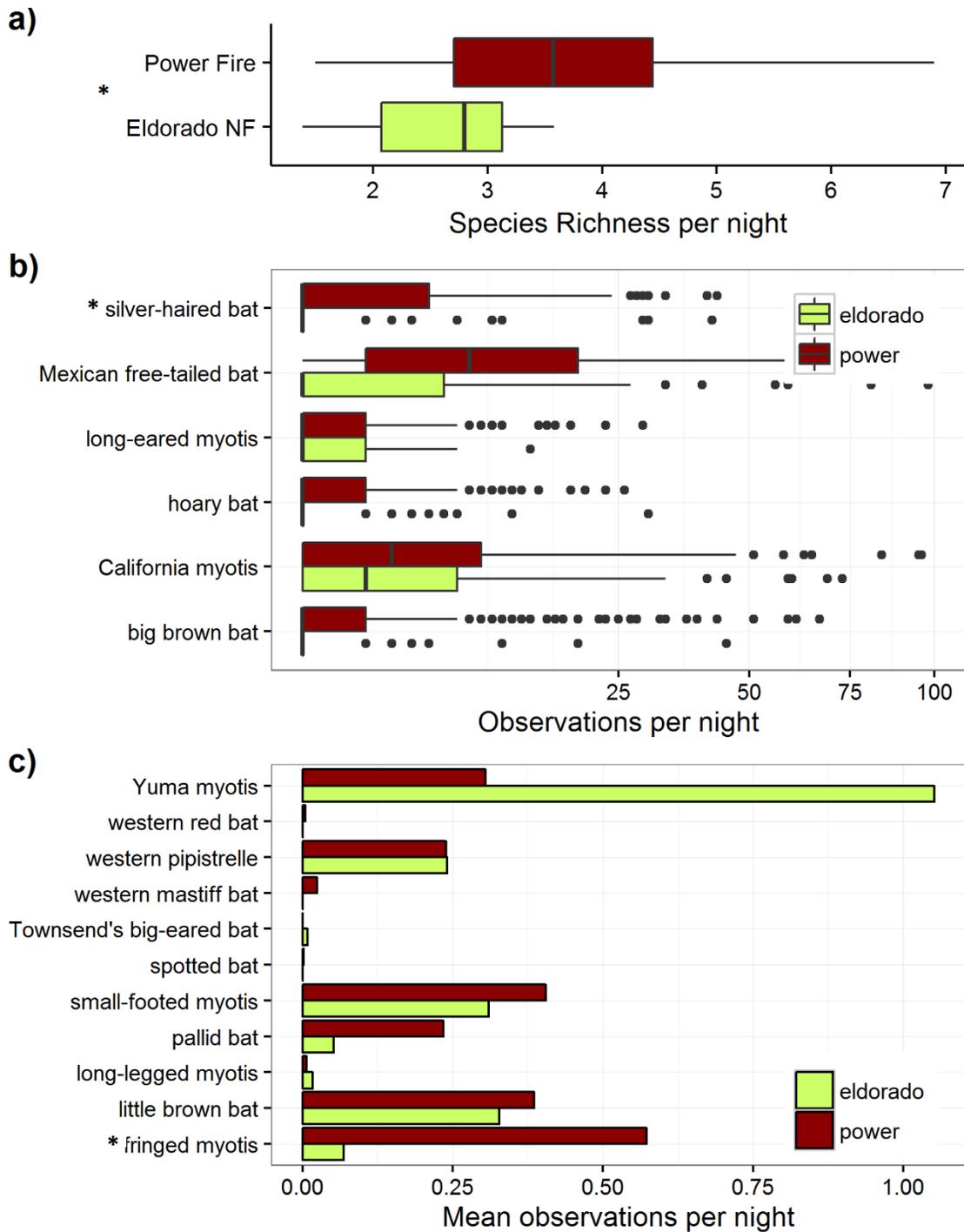


Figure 7. Observations within the Power fire burn area and the unburned Eldorado National Forest. a) Boxplots of nightly species richness. b) Boxplots of nightly observations of common species (x-axis on the log scale). c) Barplot of mean nightly observations of rare species. * indicates significant differences.

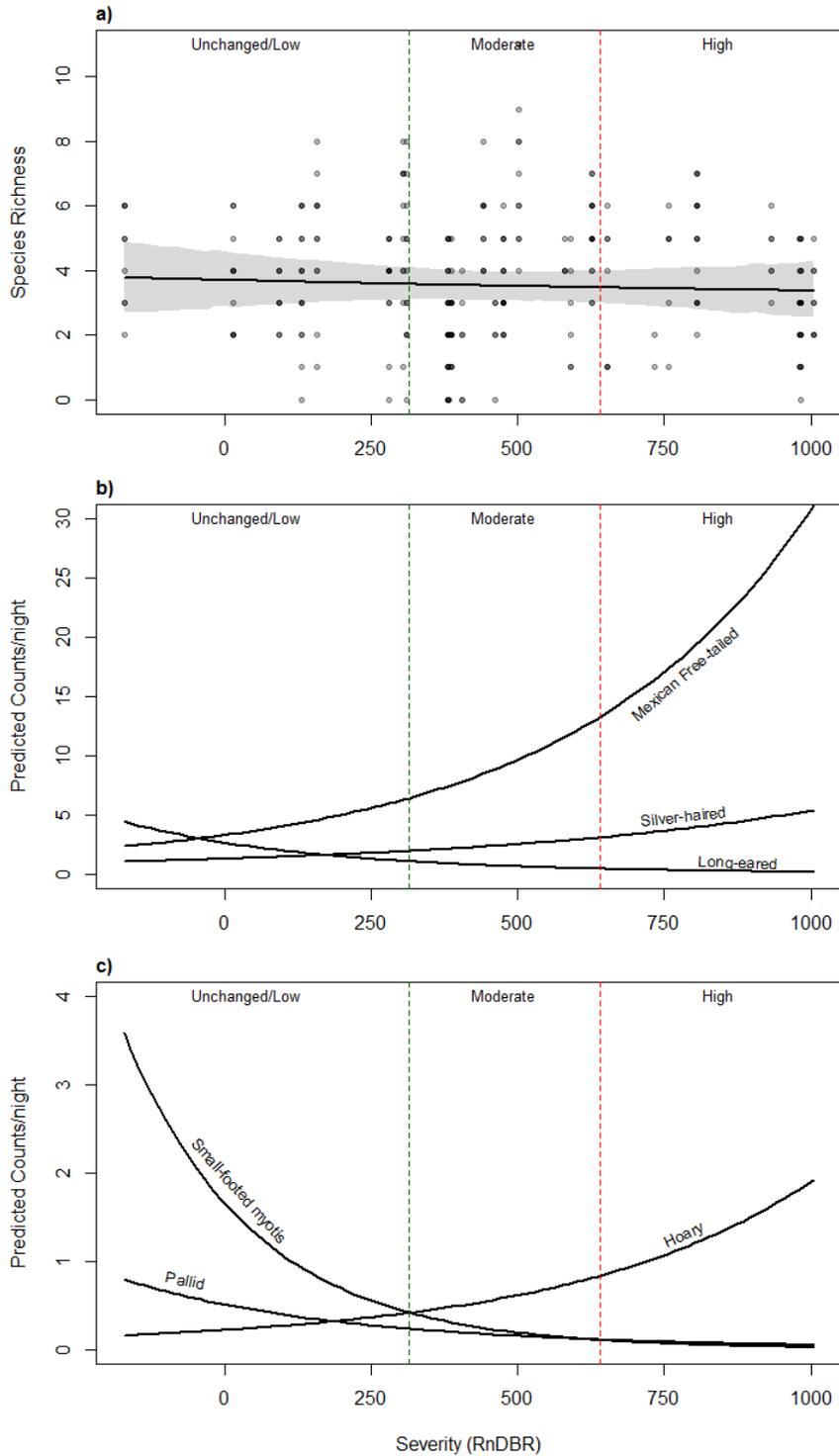


Figure 8. Modeled effect of burn severity on a) species richness, b) three common species, and c) three rare species. Only the six individual species with significant relationships with burn severity are shown. Burn severity categories and RnDBR breaks according to Miller and Thode (2007) are also indicated.

Management effects

Within the high severity component of the Power Fire, we compared areas that had been actively managed for reforestation following the burn (9 locations³ and 100 survey nights) with areas that had been left to undergo succession without intervention (9 locations and 97 survey nights; Table 1). Between these two groups, species richness did not differ significantly. However, there is greater apparent variation of bat diversity among the unmanaged areas, as compared to managed areas, which recorded relatively consistent numbers of species across surveys (Figure 9a). This apparent difference may be a product of greater variation in successional rate and trajectory - and resulting habitat quality - in unmanaged areas. Alternatively, if salvage treatments were placed in relatively uniform habitat (e.g. with similar topography, snag basal area etc.), such factors not explicitly accounted for in our models may explain the similar observations across these sites. For individual species, activity levels were observed at noticeably different rates. For example, hoary bat was observed more often in unmanaged than managed areas, and fringed myotis was recorded more often in managed than unmanaged areas (Figure 9c). However, the effect of management was significant only for hoary bat (Appendix A). Hoary bat predominantly roosts in the branches of live trees often along the edges of clearings (Shump and Shump 1982). Thus, we might expect habitat heterogeneity to be associated with activity levels of this species. If reforestation activities have led to homogenization of sampled areas this might explain the reduced activity levels, but this was not directed tested here.

A number of species are known to use cavities, crevices, and exfoliating bark of snags for roosting purposes (Barclay and Kurta 2007). Theoretically, removal of snags via salvage logging would reduce the availability of roosting habitat for these species. We did not observe clear effects of salvage logging on these species in the Power Fire, with the exception of the hoary bat, which roosts in the foliage of live trees rather than snags (Shump and Shump 1982). This may indicate that sufficient roosting habitat exists across the landscape, that remaining snags in unmanaged habitats are poor roosting habitat, or that our sampling was insufficient to detect a true effect. Bats prefer snags in early stages of decay (e.g., when exfoliating bark remains on the snag bole), which supports the possibility that any negative effects of salvage logging are only evident during the first few years following a fire. Additionally, the sample size available for this analysis is relatively modest (9 unique locations in each group), and as a consequence our power to detect marginal differences is low. In summary, these results reveal little to no effect of salvage logging and reforestation on bat activity 10-12 years after the Power Fire. If effects of management exist, additional surveys and/or monitoring in the years immediately following interventions (preferably using a Before-After, Control-Impact design) will be necessary.

³ Includes one point representing 7 survey nights located in a moderate severity area but which was salvage logged and replanted.

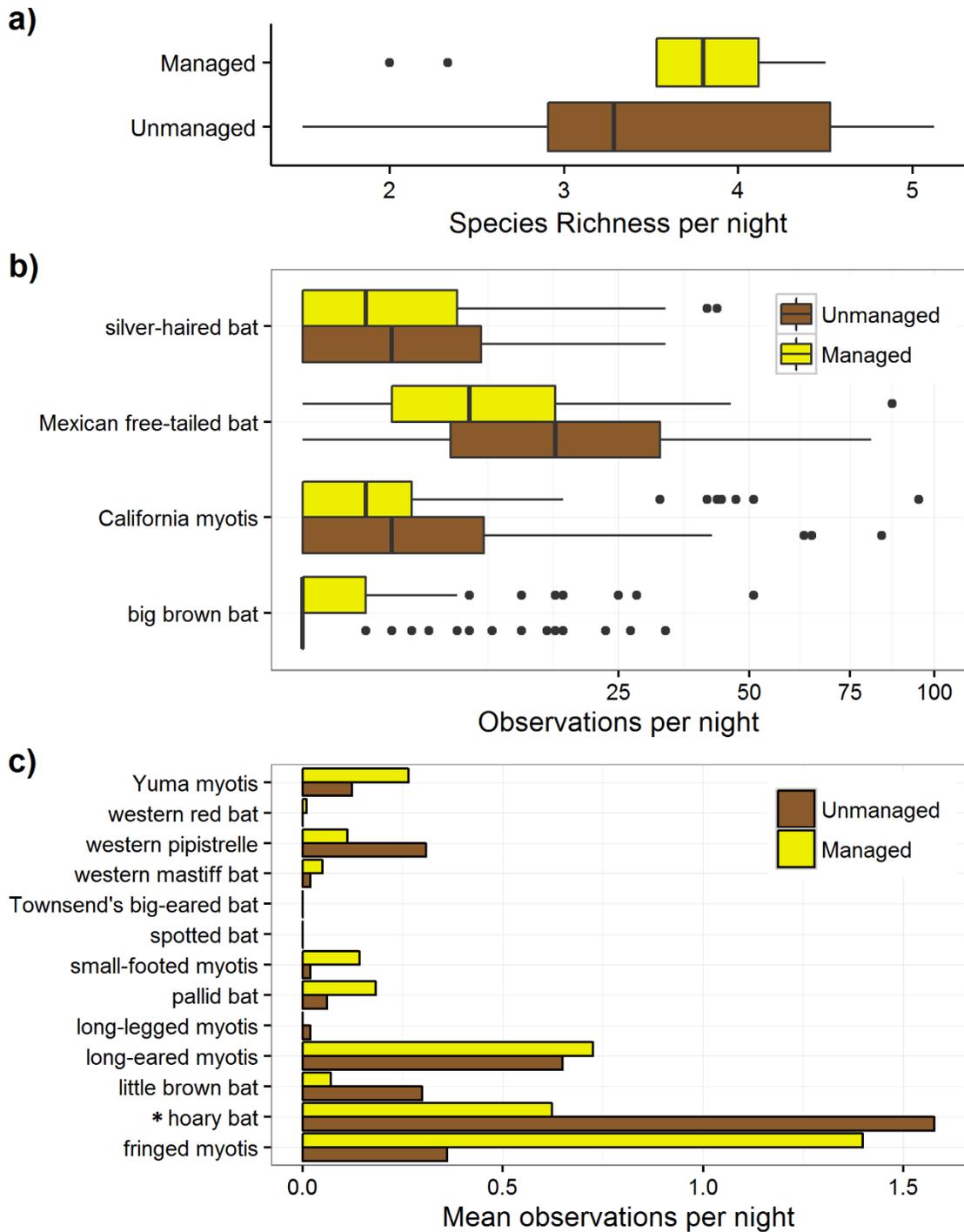


Figure 9. Observations within the Power fire high severity areas that were managed (salvaged and replanted) and not-managed following the fire. a) Boxplots of nightly species richness. b) Boxplots of nightly observations of common species (x-axis on the log scale). c) Barplots of mean nightly observations of rare species. * indicates significant differences.

Acoustic monitoring considerations and future directions

During the 2014-16 monitoring seasons, automated recording units detected 50,088 bat passes in total. Of these detections 22,427 passes (45%) were confidently classified by the Sonobat software, meaning a slight majority of individual passes could not be identified to the species level. The classification process is intentionally conservative, opting for high rates of unclassified recordings so as to avoid misclassifications and false-positive species detections. False-positives are potentially more problematic than false-negatives, and can lead to biased effects estimates when modeling species relationships with environmental variables (Clement et al. 2014). We conducted a manual review of a subset of survey nights to further elucidate the advantages and drawbacks of this automated classification approach. Among those survey nights reviewed, we found high agreement among classified recordings for most species; meaning false-positive rates were low. However, in many cases a human observer was able to affirm the presence of a bat species when the software discarded the recording due to poor quality or ambiguity. As a result, the estimates of species richness and bat activity presented in this report should be interpreted as conservative, with true values likely exceeding those observed. Likewise, effects estimates should be considered relative rather than absolute. For example, we show that Mexican free-tailed bat activity increases from less than five passes per night to approximately 30 passes per night across the range of burn severity in the Power Fire (Figure 8b). It's likely that the true Mexican free-tailed bat pass rate is higher than what was observed across the severity spectrum, but that the modeled proportional change remains a reasonable estimate of the effect of burn severity.

In addition to low classification rates among recorded passes, a species may be present but not detected at all. For example, pallid bat often hunts using auditory cues to find its prey without emitting echolocation calls (Reid 2006). Similarly, the echolocation calls of Townsend's big-eared bats are relatively quiet (Lacki et al. 2007), which means individuals must fly closer to microphones than other species to be detected. Imperfect detection is a common problem when surveying mobile and inconspicuous wildlife. For some applications, detection rates can be statistically accounted for using a family of hierarchical models often referred to as occupancy models (MacKenzie et al. 2003). This type of modeling performs well with binary presence/absence data, but models of count data such as species number or activity rates that account for detectability are still being improved (Dennis et al. 2015), and are not implemented here. In future analyses of these data we plan to assess effects on bat presence/absence in place of, or in addition to, activity levels, and will use the occupancy-modeling approach.

We will continue bat monitoring within the Power Fire through the 2017 season, adding to the data collected over the past three years. Monitoring for the subsequent season will be re-designed to focus on areas where future reforestation projects are planned. Utilizing vegetation data also collected at our survey locations, we will model bat associations with habitat variables that will be manipulated directly by management actions. For example, we will evaluate how reducing shrub cover and increasing conifer basal area is likely to influence bat occupancy over the long-term across the Power Fire project area. Additionally, we plan to explore whether proximity to important landscape features such

as riparian areas influences the bat community, with the goal of informing the spatial prioritization of management actions. For additional analyses of general habitat associations we also plan to incorporate data from other projects conducted in parallel with the Power Fire monitoring. Along with our partners at Point Blue Conservation Science, we monitored bats in and around three other fires in the central and northern Sierra Nevada. These additional fires include the 2000 Storrie and 2012 Chips fires on the Lassen and Plumas National Forests, as well as the 2013 Rim Fire on the Stanislaus National Forest.

Conclusions and Management Considerations

Wildfire is a dominant driver of forest pattern and function in the Sierra Nevada (Sugihara et al. 2006), which has profound implications for species diversity, and the geographic distributions for many taxa (Kelly and Brotons 2017; van Mantgem et al. 2015). Important work has been done assessing the impacts of wildfire on some well-studied taxa (i.e. birds and some small mammals; Fontaine et al. 2012), but up to now few studies have focused on bat community associations with burned areas and post-fire forest management (but see Buchalski et al. 2013). Thus, the bat monitoring in and near the Power Fire burn area represents an important opportunity to advance our knowledge of bats in this system, and better inform wildlife and forest management in the region.

The suitability of forest ecosystems for bat species can be characterized by the abundance of roost sites, the amount of clutter, availability of prey, and availability of water (Hayes and Loeb 2007). In this context, clutter is roughly defined as the difficulty of negotiating vegetation structure while foraging, and is related to vegetation density and structural complexity. Wildfire and forest management have the potential to influence the quality of these resources for bats through manipulation of forest vegetation. Much of the published literature as well as the results presented here are most relevant to how these manipulations affect roost availability and foraging habitat quality as related to vegetation clutter.

Bats use a variety of structures for roosting in forest ecosystems. These structures include relatively permanent natural features such as rocky outcroppings, cliffs or caves, and human infrastructure such as bridges, buildings or mines. Additionally, 13 of the 17 bat species surveyed, including the three forest service sensitive species, are known to roost in live or dead trees (snags) at some point during their lifecycle. These include the pallid bat, Townsend's big-eared bat, spotted bat, silver-haired bat, western red bat, hoary bat, California myotis, long-eared myotis, fringed myotis, long-legged myotis, and Yuma myotis (Lacki et al. 2007). Some species including the hoary bat will roost in the foliage of live trees, while many others will utilize features of snags such as cavities, crevices, exfoliating bark, and abandoned woodpecker holes, or defects of living trees. Roost trees tend to be tall, large in diameter, and located in stands with an open canopy, high density of snags (Ruppell et al. 2005), and near water and riparian areas (Brigham 2007).

High severity fire creates high densities of snags, and effectively high densities of potential roosting sites for many species. Therefore, salvage logging operations that remove large-diameter snags may reduce the availability of high quality roosting sites. However, similar to cavity-nesting birds (Saab et al. 2004), time since fire and the decay level of snags may influence the suitability of existing snags as roosting habitat (Hayes and Loeb 2007). Within the Power Fire, we observed hoary bats more often in unmanaged than reforested high severity areas, but otherwise no clear effects of past salvage logging was evident (Figure 9). If salvage logging negatively affects snag roosting bat species, it may be more evident in the years immediately following a fire when snags are less decayed. Management activities that promote the persistence, and future creation of large diameter live trees and snags would benefit many forest bat species that utilize trees as roosts (Barclay and Kurta 2007; Hayes and Loeb 2007). In the short term, retention of large trees and snags would help maintain existing roost sites. Long-term management strategies that promote mature and multi-aged forests both at the stand and landscape-scale would ensure roosting sites are continuously available into the future.

Low severity fire, either as a result of wildfire or prescribed burns, is unlikely to create many roosting sites for bats since few large trees are killed during these events. On the other hand, low and moderate severity burns reduce the clutter of forest environments by removing small-diameter trees and understory vegetation, which can improve foraging habitat for many species. Note that it is also possible that fire might consume pre-existing large snags, decreasing roosting site availability, but this would be more likely at higher fire severities and would be usually offset by snag creation (Hayes and Loeb 2007). Bat species vary in size and wing morphology, characteristics that affect flight speed and maneuverability. Small-bodied bats with low wing-loading (body mass / wing area) are able to hunt effectively in cluttered environments such as dense, closed-canopy forests, while large bats with high wing-loading are observed foraging more often in open forests or clearings (Lacki et al. 2007a). Thus, we might expect some species to be excluded from very dense forests, which is supported by the observed elevated rates of species diversity, and the activity levels of some species within the Power Fire as compared to the more cluttered surrounding habitats. Likewise, within the Power Fire, we observed compositional changes in the bat community along the gradient of burn severity, with some large-bodied species preferring the more open habitats created by high severity fire, and others occurring more often in the structurally variable low and moderate severity areas (Figure 8). This indicates that fire is an important ecological process for a bat community adapted to an ecosystem historically characterized by frequent fire and heterogeneous landscapes.

Our results suggest that the bat community responds positively to the full range of burn severities, likely due to variable stand densities and variable levels of clutter. Additionally, the availability and continued creation of large snags is likely important for roosting habitat of many species. Due to a century of fire suppression, much of the area of mixed conifer forest in the Sierra Nevada supports stands that are denser than they were in the past. The open forests found in low and moderate severity areas of the Power Fire are thus relatively rare on the landscape, and especially valuable for the bat

community. Likewise, the forest clearings created by high severity fire are important foraging areas for some species. Managing high severity areas for long-term habitat heterogeneity through the maintenance of some shrub patches where large snags remain, and promoting the retention of open stands where reforestation efforts are implemented, would most benefit the bat community of the Eldorado National Forest.

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Appendices

Appendix A – Classification and effects tables

Table A.1 - Counts of classified bat passes by species

Table A.2 - Effects estimates of fire on bat species activity

Table A.3 - Effects estimates of burn severity on bat species activity

Table A.4 - Effects estimates of management on bat species activity

Appendix B – Species Presence Maps

Appendix C – Deployment Protocol

Table A.1. Number of classified passes (corrected counts) by species for each survey point. The number of survey nights is also listed as survey effort. Species codes are used here; for common and scientific names see Table 1.

Point ID	nights	myyu	myca	myci	myvo	mylu	pahe	labl	myev	anpa	epfu	lano	myth	fabr	laci	coto	euma	eupe
EL06B.E	11	1	281	6	0	0	0	0	2	0	22	1	1	29	0	0	0	0
EL06B.W	19	0	228	1	0	0	1	0	4	0	4	6	0	70	6	0	0	0
EL10A.E	12	3	109	3	0	1	19	0	35	0	5	0	0	80	0	0	0	0
EL10A.W	8	3	4	0	0	1	0	0	1	0	0	1351	3	981	49	0	0	0
EL12B.N	8	2	72	26	0	0	1	0	9	0	15	0	0	2	7	0	0	0
EL12C.N	13	1	12	0	0	1	0	0	0	0	0	6	0	19	10	1	0	0
EL24B.W	14	111	188	0	1	26	5	0	14	3	68	12	0	2	1	0	0	0
EL30A.W	16	1	35	0	1	9	2	0	7	3	3	84	4	219	9	0	0	0
EL30B.N	15	0	32	0	0	0	0	0	1	0	0	0	0	138	25	0	0	0
PW01.2	11	0	5	1	0	0	1	0	0	0	8	31	1	279	13	0	0	0
PW01.7	7	1	6	0	0	4	0	0	0	0	1	4	0	103	3	0	0	0
PW03.1	10	6	278	20	0	10	6	0	32	1	129	104	0	198	56	0	0	0
PW03.6	10	0	265	11	0	0	6	0	2	0	3	5	4	13	0	0	0	0
PW04.6	14	0	4	1	0	0	1	0	8	0	2	86	1	1951	1	0	0	0
PW05.10	11	2	54	0	0	0	3	0	3	0	72	3	5	83	9	0	0	0
PW05.3	4	0	1	0	0	0	1	0	0	0	5	25	2	110	5	0	0	0
PW05.5	2	0	0	0	0	0	0	0	0	0	0	0	1	0	4	0	0	0
PW06.4	14	0	28	0	0	0	2	0	1	13	17	139	0	403	6	0	0	0
PW06.8	6	0	62	0	0	0	17	0	1	0	78	90	0	233	0	0	0	0

Point ID	nights	myyu	myca	myci	myvo	mylu	pahe	labl	myev	anpa	epfu	lano	myth	tabr	laci	coto	euma	eupe
PW07.5	6	0	9	1	0	1	0	0	4	0	1	23	1	75	1	0	0	0
PW07.7	14	0	7	1	0	0	0	0	3	0	16	1	4	49	0	0	0	0
PW07.9	19	1	3	0	1	4	5	0	1	2	1	47	3	742	38	0	0	0
PW08.10	9	1	110	0	0	2	3	0	8	0	12	38	10	126	4	0	0	1
PW08.4	8	0	97	0	0	0	0	0	3	0	4	8	4	148	4	0	0	0
PW08.8	7	1	400	0	0	2	3	0	32	2	4	46	92	36	9	0	0	0
PW09.10	7	1	15	0	0	0	0	0	4	3	5	84	0	90	11	0	0	0
PW09.5	17	1	25	1	0	0	2	0	3	8	8	65	36	211	5	0	0	5
PW09.7	16	0	396	1	0	2	2	0	36	1	124	70	12	506	29	0	0	1
PW10.3	10	22	60	0	0	3	0	1	10	0	349	15	0	700	4	0	0	0
PW10.5	6	0	0	0	0	0	0	0	6	0	6	2	0	32	3	0	0	0
PW10.9	12	41	108	0	0	20	2	0	12	0	2	40	4	986	97	0	0	0
PW12.10	10	9	581	0	0	9	10	1	7	25	15	82	14	28	0	0	0	0
PW12.3	9	0	58	1	0	0	2	0	45	3	1	2	11	17	4	0	1	2
PW12.5	13	10	125	0	0	1	4	0	1	2	0	43	5	43	7	0	0	0
PW13.10	12	0	47	4	0	0	0	0	76	0	11	7	9	89	2	0	0	0
PW13.2	17	0	7	4	0	0	5	0	63	0	207	44	1	480	2	0	0	1
PW14.2	16	10	123	0	0	0	2	0	2	0	1	26	1	94	6	0	0	0
PW14.5	13	0	40	1	0	4	1	0	0	0	11	29	3	242	21	0	0	0
PW15.2	12	0	4	0	0	0	1	0	1	6	1	77	1	580	5	0	0	0
PW15.4	5	0	1	0	0	0	0	0	8	0	0	11	0	41	4	0	0	0

Point ID	nights	myyu	myca	myci	myvo	mylu	pahe	labl	myev	anpa	epfu	lano	myth	tabr	laci	coto	euma	eupe
PW15.6	16	2	84	4	1	40	1	0	13	27	29	13	0	25	2	0	0	0
PW16.1	15	0	31	1	0	1	0	0	9	1	2	116	1	183	24	0	0	0
PW16.6	11	4	36	2	0	20	1	0	7	1	25	10	4	53	4	0	0	0
PW16.8	11	4	546	102	0	14	14	0	22	0	243	12	1	65	1	0	0	0
PW17.1	19	9	107	0	1	21	2	0	10	0	1	123	4	220	50	0	0	0
PW17.3	6	0	200	10	0	0	1	0	1	1	0	50	0	10	3	0	0	0
PW17.5	5	0	19	0	0	0	0	0	2	0	0	7	0	2	1	0	0	0

Table A.2. Modeled effect of fire on bat species activity. Estimates should be interpreted as the expected change from unburned to burned sites on average. Estimate and measures of uncertainty are on the log scale. Townsend’s big-eared, spotted bat, and western red bat had too few observations to be modeled.

Species	Estimate	StDev	Lower95	Upper95
pallid bat	1.24	1.10	-0.91	3.39
big brown bat	0.82	0.78	-0.71	2.35
western mastiff bat	22.46	186123.99	-364773.86	364818.79
hoary bat	0.16	0.53	-0.88	1.19
silver-haired bat*	1.49	0.64	0.23	2.74
California myotis	-0.39	0.63	-1.62	0.83
small-footed myotis	-0.24	1.07	-2.34	1.85
long-eared myotis	0.49	0.57	-0.63	1.61
little brown bat	0.04	1.00	-1.92	1.99
fringed myotis*	1.93	0.81	0.33	3.52
long-legged myotis	-0.86	0.91	-2.65	0.93
Yuma myotis	-0.96	0.89	-2.71	0.79
western pipistrelle	0.38	0.59	-0.78	1.55
Mexican free-tailed bat	0.99	0.54	-0.07	2.06

* Species for which the 95% confidence interval of the effect estimate does not encompass zero.

Table A.3. Modeled effect of burn severity on bat species activity. Estimates should be interpreted as the expected divergence from a site with an average burn severity within our sample. Estimate and measures of uncertainty are on the log scale. Townsend’s big-eared, spotted bat, and western red bat had too few observations to be modeled.

Species	Estimate	StDev	Lower95	Upper95
pallid bat*	-0.71	0.22	-1.15	-0.29
big brown bat	-0.21	0.18	-0.56	0.14
western mastiff bat	-0.56	0.45	-1.44	0.31
hoary bat*	0.66	0.19	0.28	1.02
silver-haired bat*	0.44	0.18	0.09	0.77
California myotis	-0.30	0.18	-0.65	0.03
small-footed myotis*	-1.29	0.21	-1.70	-0.90
long-eared myotis*	-0.85	0.19	-1.21	-0.48
little brown bat	-0.41	0.20	-0.78	0.01
fringed myotis	-0.19	0.21	-0.58	0.23
long-legged myotis	0.04	0.51	-1.02	0.96
Yuma myotis	-0.12	0.23	-0.57	0.31
western pipistrelle	0.00	0.21	-0.41	0.41
Mexican free-tailed bat*	0.68	0.17	0.32	1.00

* Species for which the 95% confidence interval of the effect estimate does not encompass zero.

Table A.4. Modeled effect of past management (salvage and replant) on bat species activity in high severity areas. Estimates should be interpreted as the expected change from unmanaged to managed sites on average. Estimate and measures of uncertainty are on the log scale. Townsend’s big-eared, spotted bat, and western red bat had too few observations to be modeled.

Species	Estimate	StDev	Lower95	Upper95
pallid bat	0.97	0.83	-0.65	2.60
big brown bat	-0.33	1.01	-2.31	1.66
western mastiff bat	-0.15	1.84	-3.76	3.46
hoary bat*	-0.83	0.39	-1.59	-0.07
silver-haired bat	0.03	0.46	-0.86	0.92
California myotis	0.12	0.85	-1.54	1.78
small-footed myotis	1.76	1.18	-0.54	4.06
long-eared myotis	0.41	0.59	-0.74	1.57
little brown bat	-0.79	0.84	-2.43	0.86
fringed myotis	0.13	0.90	-1.64	1.89
long-legged myotis	-33.03	6779018.97	-13286666	13286600
Yuma myotis	0.33	1.11	-1.84	2.50
western pipistrelle	-0.75	0.69	-2.11	0.61
Mexican free-tailed bat	0.06	0.58	-1.08	1.20

* Species for which the 95% confidence interval of the effect estimate does not encompass zero.