Point Blue Report

Sierra Nevada National Forests Avian Management Indicator Species

2017 Annual Report

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L. Jay Roberts and Ryan D. Burnett

3820 Cypress Drive, #11 Petaluma, CA 94954
T 707.781.2555 | F 707.765.1685
pointblue.org

Sierra Nevada National Forests Avian Management Indicator Species Project

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Table of Contents

SUMMARY

2017 marked the ninth year of monitoring four avian Management Indicator Species (MIS) across 10 National Forest units in the Sierra Nevada planning area. In 2017, we used multi-species point counts to sample a 50% subset of the 474 transects in upland habitat for Fox Sparrow, Hairy Woodpecker, and Mountain Quail and the 96 transects in riparian habitats for Yellow Warbler. The 50% subset was a response to reductions in available funding, and the total number of field locations visited was 262 upland and 68 riparian transects, slightly exceeding the 50% sampling target.

We investigated MIS distribution trends using occupancy models incorporating a large set of covariates on occupancy and detection probability with data from 2010 – 2017, excluding the pilot year of 2009. Occupancy trends indicate that Fox Sparrow occupied areas have declined over the 8 years of sampling, while Mountain Quail and Yellow Warbler distributions may have increased, and Hairy Woodpecker distribution is stable.

The value of this bioregional monitoring program goes well beyond analyzing trends for the four MIS that are the targets of this study. In recent publications we have demonstrated the importance of green forest to Black-backed Woodpecker (Fogg et al. 2014), assessed habitat associations for meadow birds (Campos et al. 2014), identified baseline reference locations for post-fire analyses in Lassen, Plumas, Eldorado, and Stanislaus National Forests (Fogg et al. 2015), provided control sample data for a BACI analysis of timber salvage in the Chips and Storrie fires (Campos and Burnett 2015), and provided control unburned sample data for post-fire monitoring analyses for birds and bats (Campos and Burnett 2016, Campos et al. 2017). We are currently investigating the long-term effects of fire and management activities on the avian community across our study area, and have submitted a manuscript for publication that assesses the effects of widespread tree mortality in the southern forests in relation to the effects of higher temperatures and water deficit that resulted from the recent drought (Roberts et al. In Review). We also provided data and assistance to other studies in preparation, including a manuscript assessing complex bird responses to time and burn severity in post-fire habitat (Taillie et al. In Review), analysis assistance to a UC-Davis student (Zack Steel) for a study exploring the response of birds to patch size of high severity fires, and our field survey locations have been used in monitoring and inventory of bats in the Power Fire (Steel and Safford 2017). We plan to continue using the Sierra Nevada bioregional monitoring dataset to help inform adaptive habitat management across this ecologically important mountain range during a period of rapid environmental change.

INTRODUCTION

In 1982, planning regulations for National Forests in the Sierra Nevada region guided the establishment of Management Indicator Species (MIS) that were chosen to reflect the diversity of plant and animal communities and their response to forest plan implementation [1982: 36 CFR 219.19(a)]. In 2007 the land management plans for each of the nine forests in the Sierra Nevada and the Lake Tahoe Basin Management Unit were amended to adopt a common suite of MIS (USDA Forest Service 2007). We developed a monitoring program to track trends in the distribution of four of these species at the bioregional scale on managed National Forest lands (Roberts et al. 2011). The four MIS targeted for monitoring with this project are Mountain Quail (*Oreortyx pictus*), Hairy Woodpecker (*Picoides villosus*), Yellow Warbler (*Setophaga petechia*), and Fox Sparrow (*Passerella iliaca*). Mountain Quail was chosen as the indicator for early and mid-seral conifer forest, Hairy Woodpecker as the indicator for snags in green forest, Yellow Warbler as the indicator for riparian habitat, and Fox Sparrow for shrub and chaparral. The total area targeted for monitoring these species encompasses approximately half of the 12 million acres of Sierra Nevada National Forest lands.

In this report we describe the field efforts and results updated through the 2017 field season including updated information on the trends in occupancy for the four MIS and Black-backed Woodpecker in unburned forest.

METHODS

Sampling Design

We conducted surveys across nine National Forests and the Lake Tahoe Basin Management Unit in the Sierra Nevada Forest Planning area (USDA Forest Service 2004). This area extends from Modoc National Forest near the Oregon border to Sequoia National Forest east of Bakersfield. Sample locations ranged in elevation from 800 – 2800 m, were limited to areas within 1 km of accessible roads, slopes less than 35 degrees, and were targeted towards green forest, shrub, and riparian habitats. These stratifications reduced potential sampling locations to approximately 50% of the area within Sierra Nevada National Forest jurisdictional boundaries (approximately 1.5 million hectares). All spatial data were processed in ArcGIS (ESRI 2011).

To ensure that our monitoring program is efficient and representative of the actively managed Forest Service land in the Sierra Nevada region - as well as within each individual forest - we used a spatially balanced sampling design (Stevens and Olsen 2004). Our goal was to ensure that our sampling design provides parameter estimates that are statistically sound (i.e. unbiased and precise) and applicable to populations across the entire region, while at the same time being flexible enough to adapt to logistical constraints as well as potential changes in effort across years due to varying levels of funding that are common to long-term monitoring projects. To achieve all this, we used a generalized random-tessellation stratified (GRTS) sampling scheme to distribute transects evenly across the region to avoid clustering in any given area (one particular forest for example) while remaining random at the local level to avoid bias due to natural spatial patterns of habitat and physiognomic conditions (Theobald et al. 2007). The spatial pattern of GRTS samples are therefore both balanced (at large scales, in this case the entire study area) and random (at small scales, in this case at approximately the National Forest Ranger District scale).

GRTS is an efficient design for monitoring programs aimed at identifying trends of species with widely differing population metrics (Carlson and Schmiegelow 2002). Another feature of GRTS is that survey locations are ordered such that any consecutive group of survey locations retains the overall spatial balance, allowing for easy adjustment to the number of locations surveyed each year (for example, due to different sizes of field crews between years) while maintaining the statistical rigor and minimizing the variance of the sample (Stevens and Olsen 2004).

The set of potential survey locations was built from a tessellation generated in ArcGIS (ESRI 2011) consisting of a grid of cells with a random origin covering the entire study area. We did not choose to stratify by geographical location (e.g. latitude bands) or by jurisdictional boundaries other than Forest Service ownership, nor did we define *a priori* a target number of survey locations within each National Forest. Thus, we used the GRTS algorithm to select survey locations with equal weight across the entire study area, resulting in the placement of survey locations proportionally to the amount and spatial distribution of suitable area for sampling (based on the habitats and other stratifications listed below).

We used two sampling frames to identify survey locations based on the species of interest. The target habitats for each species (see below) were identified from the Sierra Nevada Forests MIS documentation (USDA Forest Service 2007). Habitats for Hairy Woodpecker (‘green forest’), Fox Sparrow (‘chaparral’), and Mountain Quail (‘early to mid-seral conifer’) are widely distributed and relatively abundant across the Sierra Nevada landscape and overlap or integrate with each other. In contrast, riparian habitats, for which Yellow Warbler is the chosen indicator, are sparsely distributed across the landscape, often in linear patches that are not sufficiently represented by existing GIS habitat layers, and are discretely different than habitat identified for the three other species. Thus, we built a common sampling frame for Fox Sparrow, Hairy Woodpecker, and Mountain Quail, and a separate one for Yellow Warbler.

The original sample consisted of 250 upland sites covering the study area, and 50 riparian sites (Roberts et al. 2011). Prior to the 2013 field season we identified a set of upland sites that are logistically infeasible, many of which consisted of sites within Wilderness Areas, and thus we re-processed the GRTS site selection. Changes to the original sampling frame include removal of Wilderness and Roadless Areas, no maximum elevation limit, and increase of the lower elevation limit to 1000 m which slightly altered the distribution of the sampling frame. In the new GRTS site selection we included 232 of the original 250 upland sites after removing 18 sites that were not logistically feasible, and attempted to keep as many of these legacy sites as possible while maintaining a spatially balanced random GRTS sample. The new GRTS selection of sites included 221 of the original sites. The GRTS selection added 16 new sites, which resulted in a final sample of 237 upland sites. 2013 was the first field season these 16 sites were visited. Our recommendation is to visit all 237 upland sites each year, but given potential fluctuations in funding, the sample size can be adjusted by using priority numbers assigned by the GRTS algorithm.

Starting in 2017, in response to reduced funding availability, we split the full bioregional sample of 237 upland sites into two half-samples that we intend to visit every other year. The split sample conforms to a built-in GRTS ordered list to ensure a spatially balanced sample in each year while still maintaining a complete sample every two sampling periods. To reduce the logistical costs of visiting sites that are isolated from other locations and keep field observer pairs in relative proximity we switched a small number of sites from one yearly sample to the other. Approximately 26 sites were moved for this reason, and thus the spatial balance is affected slightly but we assume that the effect on sample quality is minimal since generally analyses will use all locations from both half-samples. Various analysis methods can be used to calculate occupancy at unvisited sites, which we discuss below.

At each of the upland sites there are two transects, each with five point count locations arranged such that four points are spaced at 250 m in the cardinal directions from a fifth point at the center. The adjacent upland transects are separated by 1 km between center points. A small number of transects vary slightly on this spatial arrangement due to logistical constraints. At each riparian field location we established two transects composed of four points each, at 200 – 300 m intervals in roughly linear arrangements along stream corridors or in meadows near stream corridors. Field reconnaissance has led to the replacement of some points and transects over the first two years of data collection due to inadequacy of remotely sensed data in identifying riparian habitat. The total sample consists of 474 upland transects distributed as 237 spatially balanced pairs, and 96 riparian transects distributed as 48 spatially balanced pairs. This sample size was selected to achieve the most robust sample possible in order to detect relatively small changes in occupancy of the MIS species at the entire study area scale given current funding levels. In most cases, i.e. for the relatively abundant species, this sample size is also large enough to evaluate forest-scale occupancy trends that could help inform adaptive forest management actions. There are approximately 30-90 transects per forest (150-450 point count locations), and most power analyses recommend at least 60 survey locations for reasonably precise and unbiased occupancy analyses given typical conditions (e.g. McKann et al. 2013). Given the length of time that these field survey locations have been visited, modern analysis methods will allow for the reductions and fluctuations in effort to have minimal impact on the power of occupancy and abundance analyses using these data (more on these methods below).

Avian Surveys

At each point we conducted a standardized point count survey (Ralph et al. 1995), where a single observer estimated the distance to the location of each individual bird detected within a five minute time span from a fixed location. All observers underwent an intensive, three week training period focused on bird identification and distance estimation prior to conducting surveys. Counts began at local sunrise, were completed within four hours, and did not occur in inclement weather. Laser rangefinders were used to assist in distance estimation. Each season we return to 50-80% of the sites a second time to conduct repeat surveys.

At the center point on upland transects we performed a five-minute playback survey for Hairy Woodpeckers and Mountain Quail and a six-minute playback survey for Black-backed Woodpecker (*Picoides arcticus*). Playback surveys were always conducted after all passive point count surveys for a transect were completed. All three species have large home ranges, and woodpeckers may vocalize infrequently, thus the probability of detecting them on a point count can be low. The goal of the playback survey was to increase the probability of detecting individuals that were available for sampling. For a more detailed account of sample design and survey methods see Roberts et al. (2011).

Analysis: Distribution Population Monitoring
To assess temporal patterns in species distributions we calculated occupancy using methods that estimate the proportion of locations (points or transects) occupied by correcting raw counts for probability of detection (MacKenzie et al. 2006). Uncorrected counts may be prone to bias and uncertainty due to variation in detectability between species, for example because of different singing rates or volumes. These methods incorporate the detection history over multiple visits to estimate detection probability. We assessed changes in MIS population distribution from 2010 to 2017 using the 'occu' occupancy function from the package 'unmarked' (Fiske and Chandler 2011, Kery and Chandler 2012) in program R (R Development Core team 2011). We excluded the 2009 pilot year of the study as a large number of points from that year were dropped and replaced in following years.

Occupancy and detection covariates can improve model fit and give more accurate trend estimates (MacKenzie et al. 2006). We revised the data and model structure for this report into a "stacked years" format where each site/year combination is treated as an independent sampling unit and all years are simultaneously fit using a single season occupancy model function with year as a covariate in the model to account for some of the correlated variance across locations. Previously we had used multi-season occupancy models which assess initial occupancy and then site extinctions and colonizations yearly, but one major advantage of the stacked years and single season model format is explicit accounting of changes in habitat variables across years. With a multi-season model it is difficult to incorporate these changes but with the stacked years single-season model it is straightforward. After eight years of field survey data, multiple vegetation survey records per point, shifts in climate including a major conifer mortality event, management, fires and other disturbances it was worthwhile to use the stacked data structure to accurately account for all these differences.

All species models included a set of covariates on occupancy including CWHR habitat type (Mayer and Laudenslayer 1988) calculated from field vegetation survey measurements (Roberts et al. 2011). We simplified the types into six classes: Sierra mixed conifer [includes ponderosa pine, Douglas fir, and pine-hardwood types]; eastside pine [includes Jeffrey pine]; white fir; red fir; lodgepole pine; and all remaining non-forest types [includes barren, bitterbrush, grassland, and wet meadow] were combined into "other". Physiographic covariates included latitude, elevation, 30-year average precipitation (PRISM Climate Group 2004), and southness which is a linear representation of aspect [=1-(absolute value(aspect-180)/180)] with north facing = 0.0 and south facing = 1.0.

Several other habitat measures were generated from local vegetation surveys within 50 m of each point count location using a relevé-based protocol on 1-3 occasions between 2009 and 2017 (Roberts et al. 2011). We then calculated yearly values of each habitat covariate for the years when no vegetation data were collected using the value from the nearest prior-year survey until a new habitat survey was conducted, then we averaged the values of the previous and new survey to create a single year transitional value. Habitat surveys were conducted more frequently on points where disturbances such as fire or management occurred. Tree cover, shrub cover, counts of snags greater than 10cm in diameter, counts of snags greater than 30cm in diameter, and total live tree basal area covariates were estimated using our field habitat surveys (see Roberts et al. 2011 for details of habitat survey methods). All variables were verified to be non-collinear by calculating the variance inflation factor of each variable in the dataset and all were <3.0 (Heiberger 2017). The riparian data included a slightly different set of covariates: latitude; elevation; tree cover; shrub cover; shrub height; basal area; and four ground percent cover variables - forbs, grass, litter, and barren.

Detection covariates were the same as listed above for occupancy but excluding habitat type, elevation, latitude, and also included day of year and time of day of each survey event. A factor variable for year was included in both the occupancy and detection portions of the model, as was an interaction term for latitude and elevation to account for similar weather and habitat conditions occurring at different elevation between the southern and northern ends of the Sierra Nevada region. Year was a required covariate on occupancy, but only included as a covariate on detection if it was supported by AIC model selection. A small number of missing covariates were replaced with average values across other points in the same transect, or other nearby transects.

We chose final models for each species by iteratively removing one occupancy and detection covariate with the lowest significance until AIC increased by at least two points. We then chose the lowest AIC model from among all models evaluated as the final model. Model selection results are not shown. See Roberts et al. (2013) for an analysis of MIS habitat and topographic associations. Means and confidence intervals of location-level occupancy were estimated using 100 parametric bootstraps and an empirical Bayes method with the functions 'ranef' and 'bup' (Fiske and Chandler 2011). Reported values are means, standard errors, and 95% quantiles of the occupancy estimates. We estimated occupancy at the transect scale for Hairy Woodpecker and Mountain Quail (where transect is considered an independent sampling unit) and the point scale for Fox Sparrow and Yellow Warbler (where point is considered an independent sampling unit). The scale at which we aggregated the detections varied by species in this way in order to create estimates at a similar scale to each species home range size (Mackenzie et al. 2006). This was done to ensure that occupancy estimates were closely related to other metrics of management value such as number of breeding pairs in territorial species (Mackenzie and Nichols 2004).

Since Hairy Woodpecker is the indicator for snags in *green* forest, we fit models of occupancy after removing all transects with at least one point that was located within 50m of a patch of medium to high severity fire that occurred in 1993 or later to minimize detections of birds within fire-influenced forest. We did not exclude transects outside but close to recent fires as our study is designed to monitor specific habitat types regardless of their landscape context. Thus, for a wide ranging species such as Hairy Woodpecker, a small portion of the birds detected in green forest may be at least in part using adjacent burned forest habitat. The 2017 green forest sample includes 233 of the 474 upland transects. Because chaparral and conifer habitats represent a successional continuum in much of the Sierra Nevada region, we included all upland locations in occupancy models for Fox Sparrow and Mountain Quail. The upland sample includes 2357 points on 474 transects, which in 2017 was reduced to 1297 points on 262 transects. The riparian sample includes 388 points on 96 transects, which in 2017 was reduced to 271 points on 68 transects.

Each species varies in the distance at which observers can detect typical vocalizations (e.g. songs, calls, drums) and therefore the effective area sampled varies when the distance of detections is not standardized. In the extreme case of Mountain Quail, individuals were regularly recorded at estimated distances of over 300 m, therefore a potentially sizable proportion of detections were from single individuals detected multiple times on adjacent points. We correct for these 'double counts' by limiting the detections included in all analyses to 100 m from the observer. Using this distance cut-off makes it unlikely that we included double counts of the same individual on adjacent survey locations that were at least 250 m apart and is within the effective maximum detection distance of all of the species analyzed in this report. Detections of Mountain Quail and Hairy Woodpecker from both the passive and playback surveys are included for transect-scale occupancy analyses. The playback surveys from 2010 include only the single 5-minute Hairy Woodpecker and Mountain Quail calls, but from 2011 onward we included detections from the additional 6-minute Black-backed Woodpecker call survey.

Data Management and Access: Sierra Nevada Avian Monitoring Information Network

Further results for MIS and all other species detected during MIS surveys can be found on the Sierra Nevada Avian Monitoring Information Network (SNAMIN) website (data.prbo.org/apps/snamin/). Across the eight years of this project, not including the 2009 pilot season, we have amassed over 565,000 individual bird records of more than 200 species at point count stations spread across 1.5 million hectares of National Forest land in the Sierra Nevada planning area. SNAMIN allows users to generate summary, abundance, and species richness analyses for MIS as well as all other species detected at the scale of individual transects, ranger districts, forests, or the entire bioregion. In addition to the analyses listed above, there are map tools for visualizing the spatial distribution of survey locations and presence/absence of species at those locations and a link to request raw data (data.prbo.org/apps/snamin/index.php?page=bioreg-home-page).

RESULTS

Survey Effort

In 2017, we surveyed 2266 point count stations on 330 transects (upland and riparian combined; Table 1). We conducted two survey visits at 45% of transects for a total of 479 transect visits (2010 – 2016 range = 835 – 987). We visited 68 of the 96 riparian sample transects, focusing on the sites within the sample that had a meadow component. Roughly half of the number of surveys were completed this year relative to past years due to the sampling redesign in response to reduced budgets.

MIS Occupancy Trends

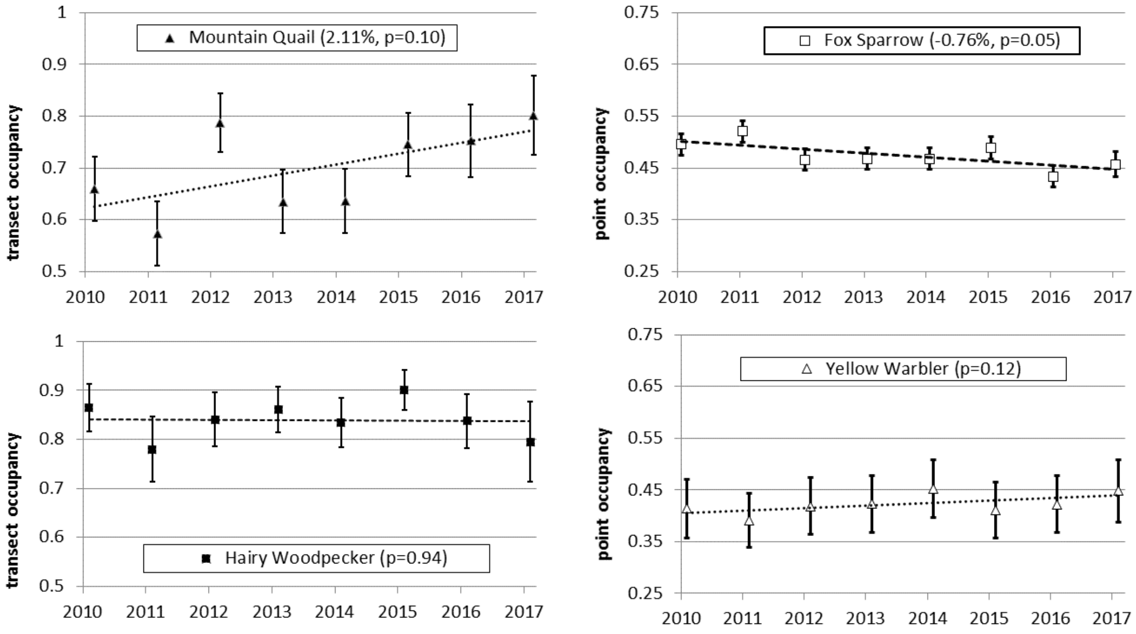
Fox Sparrows were detected at 34% of the upland point count stations we surveyed in 2017, and point scale occupancy corrected for detection probability was 0.46 (95% CI: 0.42 – 0.49). This yearly estimate is the second lowest among all years (range 0.43 – 0.52), where the highest value was in 2011 (Figure 1). The trend in Fox Sparrow occupancy averaged a statistically significant 0.76% annual decline from 2010-2017 ($P = 0.05$). Probability of detection at average covariate values was 0.68 (95% CI: 0.66 – 0.69) and was constant across years because there was no support for year as a covariate of detection.

Table 1. Survey effort by year. The full upland sample includes 474 transects, in 2017 we reduced effort by 50%. In 2009 we targeted 50 riparian transects and in 2010 and 2011 we increased the target number to 96.

		2009	**2010**	**2011**	**2012**	**2013**	**2014**	**2015**	**2016**	**2017**
Transects Visited	*upland*	415	464	472	462	473	474	474	474	262
	riparian	43	94	96	100	96	97	96	96	68
Second visits	*upland*	250	267	220	369	303	261	255	208	119
	riparian	16	65	88	56	52	55	44	72	22
Second visit rate	*upland*	60%	58%	47%	80%	64%	55%	54%	44%	45%
	riparian	37%	69%	92%	56%	54%	57%	46%	75%	44%

Hairy Woodpeckers were detected at 53% of the upland transects we surveyed in 2017, and transect scale occupancy was 0.79 (95% CI: 068 – 0.90), with a non-significant annual trend (*P* = 0.94). The estimates from 2016 and 2017 appear to be much lower than the high in 2015 of 0.90, although confidence intervals overlap (Figure 1).). Probability of detection in 2017 at average covariate values was the highest across all years at 0.56 (95% CI: 0.39 – 0.72) and ranged from a low of 0.39 in 2012.

Figure 1. Multi-year occupancy estimates from 2010 – 2017 for four upland Management Indicator Species across Sierra Nevada National Forests. Error bars show +/-1 standard error. Linear trend per year percent increase/decrease is shown in figure legends, along with significance.



Mountain Quail were detected at 33% of upland transect locations in 2017, and transect scale occupancy was 0.80 (95% CI: 0.70 – 0.89). There was weak evidence of a 2.11% annual increase (*P* = 0.10), but the yearly variation was large (Figure 1). Probability of detection in 2017 at average covariate values was 0.37 (95% CI: 0.22 – 0.54) and ranged from a low of 0.24 in 2016 to a high in 2011 of 0.44.

Yellow Warblers were detected at 39% of the riparian point locations we surveyed in 2017, and point scale occupancy was 0.45 (95% CI: 0.36 – 0.53). Occupancy in other years ranged from 0.39 – 0.45 (Figure 1) with a non significant 0.48% per year increase (*P* = 0.12). Probability of detection was 0.52 (95% CI: 0.46 – 0.58), even across all years due to year being removed during model selection.

Covariates of Occupancy

Evaluating covariate associations with MIS occupancy and detection can provide useful information on the influence of various habitat parameters on these species distribution patterns.

Eleven covariates significantly influenced Fox Sparrow occupancy (Table 2). Within our sampling frame, occupancy of Fox Sparrow increased at more southern latitudes, at higher elevations, at low small (>10cm) snag density and high large (>30cm) snag density, in areas with lower tree and higher shrub cover, low basal area, larger trees, and higher precipitation. Relative to the reference habitat type of Sierra Mixed Conifer (the most abundant type) Fox sparrow had higher occupancy in chaparral, and lower occupancy in other, eastside pine, lodgepole pine, and red fir. Occupancy was higher prior to the drought (2010, 2011) and lowest in 2016.

Table 2. Standardized coefficient estimates, unconditional standard errors, and P-value derived from the lowest AIC occupancy model for Fox Sparrow. Non-signficant (P>0.1) habitat type and year effects are not shown.

Fox Sparrow	**B**	**SE**	**P(>|z|)**
Occupancy (ψ)			
Intercept	0.07	0.08	0.38
Latitude	-0.54	0.03	0.00
Elevation	0.97	0.04	0.00
Snag Density (>10cm)	-0.10	0.04	0.01
Snag Density (>30cm)	0.19	0.05	0.00
Tree Cover	-0.16	0.04	0.00
Shrub Cover	0.24	0.03	0.00
Basal Area	-0.14	0.05	0.00
Maximum tree diameter	0.10	0.03	0.00
Precipitation	0.78	0.03	0.00
Year: 2011	0.19	0.09	0.05
Year: 2016	-0.32	0.09	0.00
Habitat: Chaparral	0.76	0.11	0.00
Habitat: Eastside Pine	-0.13	0.07	0.09
Habitat: Lodgepole Pine	-1.84	0.13	0.00
Habitat: Red Fir	-0.89	0.09	0.00
Habitat: Other	-1.18	0.24	0.00

Detection Probability (ρ)			
Intercept	0.75	0.04	0.00
Tree Cover	-0.27	0.04	0.00
Shrub Cover	0.62	0.04	0.00
Basal area	0.08	0.05	0.08
Time of day	0.03	0.02	0.12
Day of year	0.05	0.02	0.04

Six covariates were strong predictors of higher Hairy Woodpecker occupancy: higher densities of both small and large snags, low shrub cover, high basal area, southern latitudes, and low precipitation (Table 3). The only significant year effect was 2017, which was negative.

Table 3. Standardized coefficient estimates, unconditional standard errors, and P-value derived from the lowest AIC occupancy model for Hairy Woodpecker. Non-significant (P>0.1) year and habitat type effects included in final model are not shown.

Hairy Woodpecker	**B**	**SE**	**P(>|z|)**
Occupancy (ψ)			
Intercept	2.47	0.49	0.00
Latitude	-0.39	0.11	0.00
Shrub Cover	-0.50	0.12	0.00
Basal Area	0.53	0.22	0.01
Snag Density (>10cm)	0.32	0.23	0.15
Snag Density (>30cm)	0.42	0.28	0.13
Precipitation	-0.20	0.11	0.05
Year: 2017	-1.31	0.68	0.02
Detection Probability (ρ)			
Intercept	-0.27	0.13	0.04
Tree Cover	-0.09	0.04	0.04
Basal Area	-0.18	0.05	0.00
Time of day	-0.07	0.03	0.05
Year: 2014	0.35	0.17	0.04
Year: 2016	0.31	0.18	0.09
Year: 2017	0.51	0.22	0.02

Eight covariates explained variation in Mountain Quail occupancy (Table 4). Mountain Quail were more likely to occur further south, at higher elevations, and in areas with low tree cover but large tree diameter, with few large snags, and low precipitation. Significant effects of habitat types included positive for chaparral and negative for lodgepole pine. The only significant year effect was 2012 which was positive.

Table 4. Standardized coefficient estimates, unconditional standard errors, and P-value derived from the lowest AIC occupancy model for Mountain Quail. Non-signficant (P>0.1) year and habitat type effects are not shown.

Mountain Quail	**B**	**SE**	**P(>|z|)**
Occupancy (ψ)			
Intercept	1.36	0.38	0.00
Latitude	-2.05	0.25	0.00
Elevation	0.56	0.18	0.00
Tree Cover	-0.70	0.13	0.00
Maximum Tree Diameter	0.30	0.15	0.05
Snag Density (>10cm)	-0.26	0.08	0.00
Precipitation	-0.22	0.12	0.06
Habitat: Chaparral	1.03	0.48	0.03
Habitat: Lodgepole Pine	-2.35	0.60	0.00
Year: 2012	1.09	0.46	0.02
Detection Probability (ρ)			
Intercept	-0.60	0.14	0.00
Shrub Cover	0.29	0.05	0.00
Basal Area	-0.24	0.05	0.00
Time of Day	-0.22	0.04	0.00
Day of Year	-0.26	0.05	0.00
Precipitation	0.49	0.07	0.00
Year: 2011	0.34	0.20	0.09
Year: 2012	-0.41	0.18	0.02
Year: 2015	-0.35	0.18	0.05
Year: 2016	-0.55	0.20	0.01

There were eight significant predictors of Yellow Warbler occupancy: lower elevations, low tree and high shrub cover, low shrub height, low basal area, and high cover of barren ground, grass, and leaf litter (Table 5). None of the year coefficients were significant.

Table 5. Standardized coefficient estimates, unconditional standard errors, and P-value derived from the lowest AIC occupancy model for Yellow Warbler. Non-significant (P>0.1) year effects are not shown.

Yellow Warbler	**β**	**SE**	**P(>|z|)**
Occupancy (ψ)			
Intercept	-0.45	0.21	0.03
Elevation	-0.58	0.07	0.00
Tree Cover	-0.19	0.10	0.05
Shrub Cover	0.17	0.08	0.04
Shrub Height	-0.63	0.12	0.00
Basal Area	-0.64	0.12	0.00
Grass Cover	0.52	0.10	0.00

Leaf litter Cover	0.36	0.09	0.00
Barren ground Cover	0.16	0.08	0.06
Detection Probability (ρ)			
Intercept	0.08	0.12	0.48
Tree Cover	0.49	0.10	0.00
Shrub Cover	0.29	0.08	0.00
Shrub Height	1.44	0.11	0.00
Basal Area	-0.56	0.14	0.00
Day of Year	-0.13	0.07	0.05
Time of day	0.24	0.06	0.00

DISCUSSION

The Sierra Nevada bioregional monitoring program tracks trends in avian MIS distributions but can also be used to track population and distribution changes over time for at least 40 other bird species, a valuable source of information to better understand the patterns of distribution for a substantial portion of the avian community of the Sierra Nevada. It can help to inform adaptive management decisions at multiple scales from the entire Sierra Nevada region down to individual forests, and help recognize changes in habitat conditions and ecological function related to overriding factors such as climate change.

Similar to some of our previous reports (Roberts et al. 2014, 2015) we have detected sizeable year to year variation in occupancy, including one trend that may indicate a consistent directional change across the eight years included in the analyses. An average annual decline in Fox Sparrow occupancy of 0.76% may be an ecologically significant pattern. A 0.42% annual increase in Yellow Warbler occupancy is not statistically significant, and the distributions of Hairy Woodpecker and Mountain Quail are even more variable across years. The 2017 Hairy Woodpecker occupancy estimate is the lowest we have recorded, and that follows a high just two years earlier in 2015. The recent high values in Mountain Quail occupancy lead to a strong positive but not statistically significant linear trend, and given the pattern across years we suspect it is not an ecologically significant population change but rather yearly variation in detectability or shifts in home range occupancy due to unmeasured factors. Thus Yellow Warbler, Mountain Quail, and Hairy woodpecker appear to be stable from a short- term (<10 years) perspective. Given the wide range of climate and weather conditions during the study period, the lack of even larger population distribution changes is notable.

MIS Summaries

During our past 8 years of monitoring, across large variation in temperature and precipitation (well above average snowpack in 2011 followed by historically severe drought in 2012-2015), it is not surprising to see fluctuations in occupancy for these four species. This past monitoring season in 2017 saw a return to dry and hot weather conditions. While we don't see obvious changes in MIS occupancy that correlate directly with weather patterns, it is important to recognize that the effects of climate on wildlife and their habitats may manifest over long timeframes, and that certain portions of the study area may be experiencing changes in ability to support MIS and other species. For example, in a manuscript currently in review we have assessed the effects of drought and bark beetle-induced pine mortality in southern Sierra Nevada for MIS and many other species (Roberts et al. In Review). Fox Sparrow and Mountain Quail both show declines in areas with high mortality and in response to higher temperatures. Hairy Woodpeckers decline in response to higher temperatures, but increase where conifer mortality is high. Yellow Warblers in shrub habitats (riparian areas were not included) appear to increase where water deficit is high and where conifer mortality is high.

With the continuation of this avian bioregional monitoring program, it will be possible to track the influence of climate-related changes on a large number of species (as well as their habitats) to help inform climate-smart adaptive management tailored to the likely changes that will occur within the Sierra ecosystem. Understanding how each of these species responds to changes on the landscape, including management activities, climate, fire, and other disturbances, can help inform ongoing management decisions. In the following species summaries we update information from previous reports to provide a summary of the patterns of the species distribution and habitat associations that may prove useful in using these species to help guide adaptive management.

Fox Sparrow

Fox Sparrow occurred at nearly 50% of all point count stations in the study area and have been detected on every National Forest unit. However, the species was not evenly distributed across the region. Fox Sparrow occupancy was considerably higher in the central and southern Sierra, and lower on the three forests dominated by east-side habitat (the Modoc, Lassen, and Inyo National Forests; Roberts et al. 2013). We detected a statistically significant but small magnitude, decline in Fox Sparrow occupancy from 2010 to 2017. While drought may be a cause of the possible decline in this species, a number of management actions can also alter habitat suitability for Fox Sparrow. These include fire suppression which results in a slow succession of chaparral and open forest into closed-canopy forest. More immediate are the short term effects of fire burning chaparral habitat, and the effects of any management activities that reduce shrub density and extent. In the Northern Sierra we found that this species continued to increase in abundance for more than 10 years following stand replacing fire where they became exceedingly abundant in the whitethorn (*Ceanothus cordulatus*) dominated chaparral in the Storrie Fire footprint (Campos and Burnett 2014). In contrast, they appeared to be declining in beetle induced conifer mortality in the southern Sierra (Roberts et al. In Review). This could be a drought response at the edge of the species' elevation range coupled with a lack of time for shrub growth in response to more open canopies. Continued monitoring will help determine the effects of conifer mortality on this shrub associated indicator species.

Hairy Woodpecker

Hairy Woodpecker occurred on nearly 90% of transects in the study area and have been detected on every National Forest unit. There has been no apparent trend in distribution over time, although 2017 occupancy was the lowest recorded thus far. They are evenly distributed across the entire region (Roberts et al. 2013) and are the most widely distributed woodpecker species in the Sierra Nevada, occurring in all conifer dominated habitat types east and west of the crest. Though they are widespread, they are not among the most abundant birds in the Sierra Nevada. Their relatively large home ranges limit high densities. Detection probability for this species is relatively low, even with playback, such that field survey counts considerably underestimate the species' true occupancy.

They are strong primary excavators and as such they play an important role as cavity creators in the Sierra Nevada and throughout their range (Martin and Eadie 1999, Tarbill 2010). The species is closely tied to snags in both green and burned forest, not only for nest sites but for foraging resources. Like most of the woodpeckers in the Sierra Nevada, their greatest density is in recently burned forest (Burnett et al. 2012), as well as low severity fire and treatments that reduce understory cover (unpubublished data). Hairy Woodpecker has also shown a consistent pattern of increased occupancy or abundance as a result of commercial thinning treatments which reduce understory cover but do not increase snag density, such as aspen restorations (Campos & Burnett 2014) and shaded fuel breaks. Our evaluation of their occupancy here excludes areas that have burned in the past 20 years, thus it is an underestimate of the species occupancy across all habitats in the region.

Mountain Quail

Mountain Quail occupancy was higher in 2012, 2015, 2016, and 2017 than in other years, and an insignificant positive linear trend in the population was detected. High inter-annual variation may be related to nesting phenology due to weather patterns in late winter and early spring causing Mountain Quail to reduce singing rates during active incubation periods. Because of their large territory size, this species' total population in the Sierra Nevada is likely far less than for species with smaller territory sizes whose occupancy has been estimated at the point scale (e.g. Fox Sparrow).

While they are the indicator for early and mid-seral conifer forest, we have found that this species is tied to shrub dominated habitats. But, unlike Fox Sparrow, Mountain Quail readily occur in the understory of mature open-canopy forest with a shrub component (Roberts et al. 2013). In our occupancy models both a positive association with chaparral habitat types and shrub cover, and a negative association with tree cover indicate conditions consistent with chaparral and early seral forest. However the positive association with maximum tree diameter indicates that a more complex mixture of age classes could increase habitat quality.

Yellow Warbler

Yellow Warblers occur at nearly 45% of our riparian point count stations and they appear to be increasing, though this trend is not statistically significant. The fact that they do not occur at up to 55% of our riparian sampling stations suggests that their distribution may be limited by a variety of factors such as elevation. Occupancy was higher at low elevation locations with high shrub cover, low tree cover, and lower tree basal area, and ground cover including grass and leaf litter.

We used data from our MIS monitoring along with several other datasets to evaluate Yellow Warbler habitat associations in the Sierra Nevada (Campos et al. 2014). In that analysis, willow cover was the single best predictor of density in riparian meadows across the Sierra Nevada. Once willow cover exceeded 40% the effect of willow cover on density increased. This suggests that managing for over 40% willow cover (within a 50 m radius) may be an important threshold for this species. Conifer tree cover was negatively associated with the species, and riparian tree cover was positive. Conifer removal from riparian areas, especially when it invigorates riparian trees and willows, should benefit this species. This same analysis found a strong positive relationship between Willow Flycatcher occupied meadows and Yellow Warbler density. High Yellow Warbler density may be a good indicator of suitable Willow Flycatcher habitat. Since endangered species with small disjointed populations may be slow to colonize restored habitat, Yellow Warbler may be a reliable indicator of whether enhancement activities achieve desired conditions for Willow Flycatcher. Riparian meadow restoration that restores floodplain function and increases the cover of willow (e.g. pond and plug) has also been shown to increase Yellow Warbler and other riparian associated avian species abundance in the Sierra Nevada (Burnett and Campos 2015).

Closing remarks

This eight year data set represents a significant investment on the part of the USFS and Point Blue, and its value may continue to increase over time as a baseline to assess the effects of environmental change into the future. The utility of bioregional monitoring programs, such as this ongoing Sierra Nevada project, goes far beyond analyzing trends for target species. In one publication, we demonstrated the importance of green forest to a burned forest specialist, the Black-backed Woodpecker (Fogg et al. 2014). Riparian meadow transects from this project were included along with locations from other studies to examine habitat associations for meadow birds (Campos et al. 2014), and we plan to integrate meadow field sites from this project into a future Sierra-wide meadows monitoring program. We have also used unburned locations as reference data for post-fire analyses on Lassen National Forest (Campos and Burnett 2014), and in an analysis of the effects of fire on the Eldorado National Forest (Fogg et al. 2015). Each year several of our field sites have burned, including over 60 locations in the 2013 Rim Fire on Stanislaus National Forest, and along with adjacent unburned locations we are currently using these data in analyses of the avian community in post-fire habitat (Taillie et al. In Review). We are also completing manuscripts evaluating the response of the avian community to the massive recent drought and tree mortality event in the southern Sierra, as well as comparing the long-term effects of fire and silvicultural treatments. We plan to continue using the Sierra Nevada bioregional monitoring dataset in future reports, publications, and presentations to help inform adaptive forest management and conservation of biodiversity across this ecologically important mountain range.

Gaps in sampling efforts may negatively influence numerous important products of this project including calculating trends, monitoring response to climate changes, and assessing responses of bird species to disturbances including fire and management. Statistical confidence in occupancy estimates was reduced in 2017 as a function of smaller sample size, however we found that the regional occupancy estimates were comparable to previous years. Planning for the future of this project should consider future goals and objectives, including those related to: adaptive management, climate change, and others.

APPENDIX A: Black-backed Woodpeckers in Green Forest

The Black-backed Woodpecker (*Picoides arcticus*) is an uncommon habitat specialist that reaches its greatest density in moderate and high severity burned forest (Hutto 2008, Saracco et al. 2011). This species also inhabits green forest throughout its range but there are relatively few studies of their life history outside of burned areas. In 2014 we published an analysis on their habitat associations and occupancy in unburned forest in the Sierra Nevada of California (Fogg et al. 2014). In this appendix we update results published in that manuscript and the previous report (Roberts et al. 2017) with one additional year of survey data.

Methods

To evaluate Black-backed Woodpecker occupancy in green forest, we used survey data from the upland sampling locations described in the body of this report (or see Roberts et al. 2011). We used point count data from 2011-2017 on the 474 upland transects located on 10 national forest units (Table A1). We selected all upland forest transects that have not been influenced by burned forest patches for use in this analysis. These transects are defined as areas that had not burned at moderate or high severity from 1993-2017 and were more than 2 km from recent moderate or high severity fire patches from 2004-2017 (*n* = 362 transects).

At each of the five point count stations within a transect we conducted a standardized unlimited distance 5 minute point count survey (Ralph et al. 1995), where a single observer estimated the distance to the location of each individual bird they detected (hereafter “passive surveys”). Following the five passive surveys, at the center point of each transect only, we conducted a 5 minute playback survey for Hairy Woodpecker (*Picoides villosus*) and Mountain Quail (*Oreortyx pictus*), and a 6 minute playback survey for Black-backed Woodpecker. We conducted surveys for the two other species as part of the MIS protocol. Black-backed Woodpecker survey duration was 6 minutes, with three increments of 25 sec playbacks followed by 95 sec of listening and watching. Playbacks included the scream-rattle-snarl and pik calls and territorial drumming sounds (recording by G. A. Keller, Macaulay Library of Natural Sounds, Cornell Laboratory of Ornithology). Playbacks were broadcast at a standardized volume (90 db) using FOXPRO® ZR2 digital game callers (FOXPRO Inc., Lewistown, Pennsylvania, USA). Playback surveys have been shown to significantly increase detection probability for this species compared to individual passive point count surveys (Saracco et al. 2011). Playback surveys were only conducted once per transect visit after all passive point count surveys were completed to avoid influencing detection probability on passive surveys via individuals drawn towards the broadcast from large distances away. The approximate range at which human observers can hear the playback calls is 200 m, but variable depending on topography and vegetation. We also included any Black-backed Woodpecker detections from the Hairy Woodpecker/Mountain Quail playback survey in our analysis dataset.

All observers underwent an intensive, three week training period focused on bird identification prior to conducting surveys. Surveys were conducted between local sunrise and 1000 h from May 13 – July 15. Surveys did not occur in inclement weather that could reduce detectability (e.g. high wind, rain, dense fog). Variable survey effort was accounted for in our occupancy modeling framework described below.

In order to evaluate Black-backed Woodpecker patterns of occurrence we used a multi-season dynamic occupancy model (MacKenzie et al. 2006) which includes probabilities of transect colonization and extinction between seasons (in our case, years). For each of the n = 362 green forest transects there were a maximum of K = 28 survey occasions (up to two visits on two separate dates each year, and two survey types per visit). We included occupancy covariates including latitude, elevation, the interaction between latitude and elevation, slope, southness, solar radiation index (Keating et al. 2007), tree cover, shrub cover, basal area, and two snag size counts (>10cm and >30cm), and were included for all parameters of the dynamic occupancy model (occupancy, probability of detection, site colonization, and site extinction).

Models were analyzed using R version 3.2 statistical software and the package ‘unmarked’ (R Development Core Team 2011; Fiske and Chandler 2011). All counts were converted to detection/non-detection (1 or 0). Both occupancy and probability of detection were fit with logit-linear models. We defined the model for occupancy probability as the logit-transformed probability of occupancy in relation to the covariates listed above. Probability of detection was modeled with an intercept term, the occupancy covariates, and a covariate for survey type, passive [0] or playback [1]. Colonization and extinction models included the occupancy covariates and included separate intercepts for each year to allow for more variance in yearly occupancy estimates. Model selection proceeded using a standard AIC stepwise removal approach where each variable with the lowest probability was removed until AIC did not improve. We derived annual occupancy estimates from the final model using the ‘smoothed’ estimator and generated standard errors for occupancy estimates using 100 non-parametric bootstraps.

Results and Discussion

Black-backed Woodpeckers have been detected on at least one green forest transect in all forests except for the Stanislaus National Forest, Sequoia National Forest, and the Lake Tahoe Basin Management Unit (Table A1). They are present on burned forest transects in Stanislaus National Forest and the Lake Tahoe Basin Management Unit, but we have not detected one on burned or unburned transects on the Sequoia National Forest (data not shown). However, other researchers have detected Black-backed Woodpeckers in the Sequoia in burned forest (Siegel et al. 2015a). At least one Black-backed Woodpecker has been detected at a total of 126 of the 362 green forest transects (35%) from 2011 to 2017.

The lowest AIC model following the stepwise removal process included five occupancy and four detection covariates (Table A2). Occupancy is positively associated with low slopes, high tree cover, high elevations, northern latitudes, and a negative interaction between elevation and latitude indicating that they occur on lower elevations in the north relative to the south. Detection probability covariates included a positive effect of >10cm snag density, and day of year (later in the season). Colonization was higher at low slopes, high elevations, high tree cover, low basal area, and low >10cm snag density and varied strongly among years with the highest in 2016 and the lowest in 2015. Extinction was higher at south facing aspects and low basal area, and also differed among years with highest in 2013, and lowest in 2014. The extinction parameter estimate was again high, as it has been in previous reports, while colonization remained relatively low. We suspect this is at least partially an artifact of the dynamic occupancy model when fitted to sparse data as shown by Miller et al. (2015), and that the rate of extinction applies to occupied sites (~25% of the sample) while the rate of colonization applies to unoccupied sites (~75%).

Estimated occupancy ranged from 0.16 in 2011 to 0.25 in 2017 with confidence intervals largely overlapping for all five years. Although the occupancy estimates are very similar to our previous analyses, the pattern among years implies a significantly increasing trend of 1.2% per year ($P = 0.01$) which aligns with our previous result (Roberts and Burnett 2017), but contradicts the potentially decreasing trend we reported following the 2014 field season (Roberts 2015).

The final model estimated colonization ranging from 0.04 (95% CI = 0.01 – 0.18) to 0.17 (95% CI = 0.07 – 0.34), varying across years. Extinction ranged from 0.19 (95% CI = 0.03 – 0.63) to 0.58 (95% CI = 0.34 – 0.80). High extinction and low colonization rates are similar to those found in burned forest in our study area (Siegel et al. 2015a). Error bars for extinction rate estimates are large, partially a result of bias due to sparse data and low initial occupancy rates confounding the estimation of site extinction with the detection process (Miller et al. 2015).

To date, all our analyses have shown that occupancy is highest in lodgepole pine and red fir forest, and is positively associated with elevation (above approximately 1500m in the northern Sierra and 1800 m in the south), northerly latitudes (more than 37 degrees), lower than 30% slopes, and patches of high snag density (> 10 snags over 10cm in diameter per acre). Though we calculated a significant positive trend in occupancy, due to the annual variation in our estimates, sensitivity of the model to new data, and relatively large error estimates we suggest that the distribution of the Black-backed Woodpecker population on actively managed unburned national forest lands in the Sierra bioregion is best characterized as either stable or slowly increasing. These results suggest once again that many of the individuals detected in green forest are not just actively dispersing across the landscape in search of burned areas, but are occupying stable home ranges. Despite the relatively large areas that have recently burned (e.g. Rim, King, and Chips Fires) during the timeframe of this monitoring project, occupancy on our unburned forest sites has not declined but rather increased. Based on their analysis of age structure from molt patterns, Siegel et al. (2015b) concluded that natal dispersal, not breeding dispersal, was the primary means by which fires are colonized by Black-backed Woodpeckers. Combined, these findings suggest that the green forest population of Black-backed Woodpeckers is not comprised primarily of birds moving through the landscape seeking recently burned areas, but could be benefitting from dispersal out of source populations in these large fires. A study of population demographics in unburned forest would be valuable for understanding the dynamics between the burned and unburned populations of Black-backed Woodpeckers in Sierra Nevada forests.

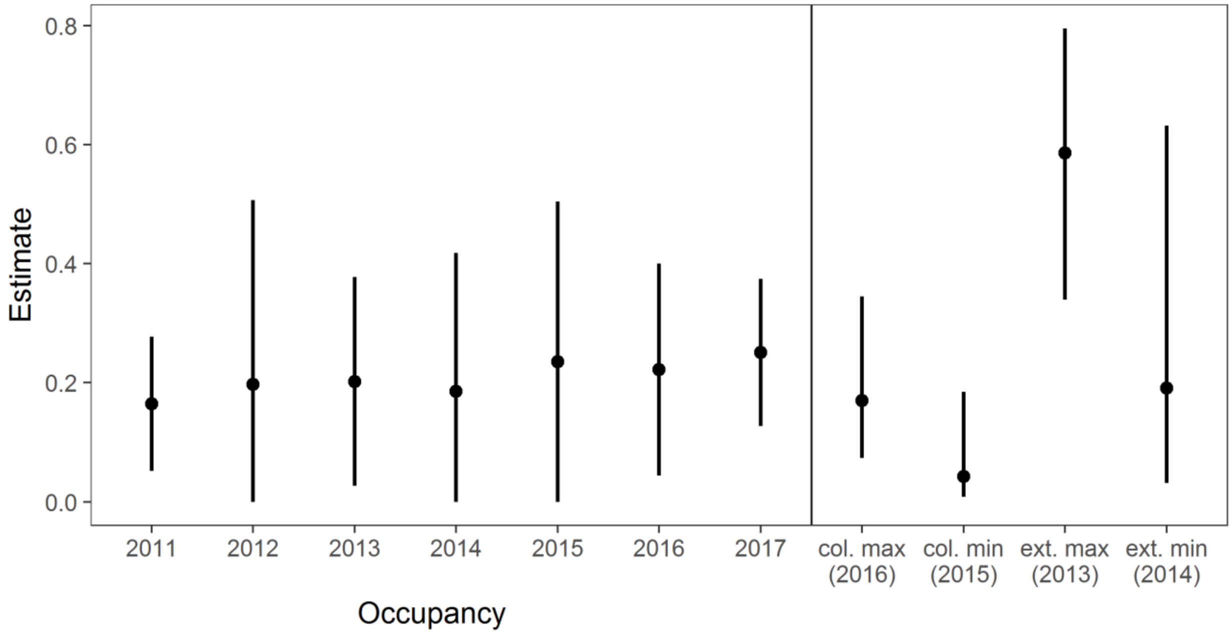
Table A1. Number of green forest transects with Black-backed Woodpecker detections for each National Forest Unit in the Sierra Nevada planning area. LTMBU = Lake Tahoe Basin Management Unit. The total number of transects surveyed for each unit is in parentheses. These data include only the transects used in the current analysis.

Forest	**2011**	**2012**	**2013**	**2014**	**2015**	**2016**	**2017**
Modoc	6 (43)	7 (43)	4 (45)	6 (45)	8 (45)	6 (45)	2 (20)
Lassen	8 (59)	11 (57)	10 (59)	7 (59)	9 (59)	7 (59)	5 (27)
Plumas	0 (38)	2 (38)	1 (38)	2 (38)	0 (38)	1 (38)	0 (20)
Tahoe	2 (40)	3 (44)	4 (44)	2 (44)	4 (44)	3 (43)	1 (21)
LTBMU	0 (2)	0 (2)	0 (2)	0 (2)	0 (2)	0 (2)	0 (0)
Eldorado	1 (37)	2 (37)	4 (40)	4 (41)	3 (41)	3 (41)	4 (19)
Stanislaus	0 (20)	4 (24)	4 (26)	0 (26)	4 (26)	0 (26)	0 (12)
Inyo	5 (17)	2 (17)	6 (19)	6 (19)	4 (19)	8 (19)	3 (13)
Sierra	5 (55)	4 (52)	6 (59)	5 (59)	6 (59)	13 (59)	4 (45)
Sequoia	0 (26)	0 (26)	0 (29)	0 (29)	0 (29)	0 (29)	0 (21)
Total	27 (337)	35 (340)	39 (361)	32 (362)	38 (362)	41 (361)	19 (198)

Table A2. Final dynamic occupancy model chosen via AIC-based stepwise variable removal.

Black-backed Woodpecker	**β**	**SE**	**P(>|z|)**
Occupancy (ψ)			
Intercept	-3.03	0.56	0.00
Elevation	1.85	0.55	0.00
Latitude	1.93	0.58	0.00
Slope	-0.44	0.30	0.14
Elevation:latitude	-0.74	0.38	0.05
Tree cover	0.41	0.26	0.11
Colonization (γ)			
Intercept 2011	-2.25	0.42	0.00
Intercept 2012	-2.72	0.50	0.00
Intercept 2013	-2.31	0.35	0.00
Intercept 2014	-2.47	0.44	0.00
Intercept 2015	-3.12	0.84	0.00
Intercept 2016	-1.59	0.48	0.00
Elevation	1.02	0.18	0.00
Slope	-0.83	0.19	0.00
Tree cover	0.30	0.18	0.09
Basal area	-0.57	0.21	0.01
Snag density (>10cm)	-0.22	0.16	0.18
Extinction (ε)			
Intercept 2011	-0.03	0.68	0.96
Intercept 2012	-0.68	0.61	0.26
Intercept 2013	0.35	0.52	0.50
Intercept 2014	-1.45	1.01	0.15
Intercept 2015	-1.15	0.71	0.11
Intercept 2016	0.31	0.79	0.69
Southness	0.57	0.23	0.01
Basal area	-0.42	0.29	0.15
Detection Probability (ρ)			
Intercept	-4.05	0.99	0.00
Slope	-0.25	0.10	0.01
Shrub cover	-0.35	0.10	0.00
Snag density (>10cm)	0.31	0.11	0.01
Day of year	0.02	0.01	0.01

Figure A1: Annual occupancy estimates for Black-backed Woodpeckers in green forest. Vertical lines bounding each point indicate 1 standard error. Colonization and extinction estimates are shown for the highest and lowest yearly values.



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