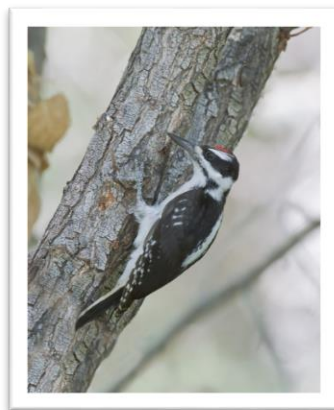
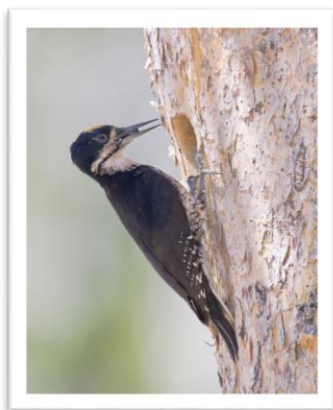


Sierra Nevada National Forests Avian Management Indicator Species



2019 Annual Report

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Sierra Nevada National Forests Avian Management Indicator Species Project

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SUMMARY

2019 marked the tenth year of monitoring four avian Management Indicator Species (MIS) across 10 National Forest units in the Sierra Nevada planning area. In 2019, 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, as well as the majority of 96 transects in montane riparian habitats for Yellow Warbler, focusing on the survey locations in meadows. The 50% subset was a response to reductions in available funding implemented prior to the 2017 field season, and the total number of field locations visited in 2019 was 250 upland and 64 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 – 2019, excluding the pilot year of 2009 and the unsampled year of 2018. Occupancy trends indicate that Fox Sparrow occupied areas have declined over the 10 years of sampling, while Yellow Warbler distribution has increased, and Mountain Quail and Hairy Woodpecker distributions are stable or possibly also increasing.

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 also published a study assessing complex bird responses to time and burn severity in post-fire habitat (Taillie et al. 2018). Recently we published a manuscript in the journal *Ecological Applications* 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. 2019). We are currently preparing a manuscript investigating the long-term effects of fire and management activities on the avian community across our study area. We also provided data and assistance to other studies in preparation including 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 2019 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 intended to visit every other year. With current funding levels we will only be surveying the “odd years” half-sample for the foreseeable future. Thus the “even years” half-sample has not been surveyed since 2016 and there are no plans in place at this time to revisit those sites until funding is available. 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 are flexible to use all locations from both half-samples, or just the odd-years half-sample. Various analysis methods can be used to calculate occupancy at unvisited sites, which we discuss below and provide data showing whether using the full versus half sample incurs any bias in MIS occupancy results.

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 and forest planning. 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 2019 using the 'occu' occupancy function from the package 'unmarked' (Fiske and Chandler 2011, Kery and Chandler 2012) in program R version 3.3 (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.

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.

Occupancy and detection covariates can improve model fit and give more accurate trend estimates (MacKenzie et al. 2006). We again used the "stacked years" data and model structure for 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. Prior to the 2017 annual report 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 ten 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 (corrected for latitudinal bias by using the residuals from a latitude*elevation linear regression), 30-year average precipitation (PRISM Climate Group 2004) which is a measure that correlates strongly with winter snowpack, and southness which is a linear representation of aspect [$=1-(\text{absolute value}(\text{aspect}-180)/180)$] with north facing = 0.0 and south facing = 1.0.

Several other habitat measures (listed below) 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). To account for vegetation changes for the years when no vegetation data were collected, we calculated yearly values of each habitat covariate 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, maximum live tree 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. We were not able to conduct any vegetation surveys in 2019 due to limitations on field crew availability, and thus there were several field locations where vegetation conditions changed but we do not have up to date data to describe these changes. We compiled notes from field visits and overlaid all points on the most recent burn severity GIS data to identify any points that had experienced fire after the 2017 and prior to the 2019 field seasons, and removed 74 points and 10 complete transects from all analyses to avoid fitting models to inaccurate vegetation data.

Covariates in the detection portion of the occupancy models were the same as listed above for occupancy but excluded habitat type, elevation, and latitude, and included day of year and time of day of each survey event. An interaction between precipitation and day of year was included for Mountain Quail to account for a pattern we identified where detectability appears to decline later in the spring, but the timing of this varies based on snow pack amount. A factor variable for year was included in both the occupancy and detection portions of the model, and was a required covariate on occupancy (i.e. not removed during model selection), but only included as a covariate on detection if it was supported by AIC. A small number of missing covariates in the occupancy models were replaced with average values across other points in the same transect, or adjacent transects. We chose final models for each species

by iteratively removing one occupancy and detection covariate with the lowest significance until AIC did not improve. We then chose the lowest AIC model from among all models evaluated as the final model. Model selection results are not shown. See also Roberts et al. (2013) for an analysis of MIS habitat and topographic associations. Means, standard errors, and 95% quantiles of location-level occupancy were estimated using 500 parametric bootstraps and an empirical Bayes method with the functions ‘ranef’ and ‘bup’ (Fiske and Chandler 2011). 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 the year 2000 or later to minimize detections of birds within fire-influenced forest. This reduced the total sample from 474 to 402 transects. 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 reduced sampling effort in 2017 lowered the green forest sample to 233 of the 402 green forest transects, and in 2019 it was 213. 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 full upland sample includes 2353 points on 474 transects, which in 2017 was reduced to 1297 points on 262 transects, and in 2019 was 1328 points on 250 transects. The full riparian sample includes 388 points on 96 transects, which in 2017 was reduced to 271 points on 68 transects, and in 2019 was 257 points on 64 transects. It is important to note that the bulk of sample size changes across years either reflected randomized (GRTS) site selection, or actual changes to landscape conditions as a result of fires and other disturbances, and thus the yearly occupancy estimates reflect a representative sample of the entire region which is appropriate for trend analyses. The exception is for Yellow Warbler, where the reduction from 96 transects in 2016 and earlier to 68 transects in 2017 and later was not random but rather based on our evaluation of whether these locations were in a meadow fed by a stream. In this report, we show results and trend estimates only for the reduced 68 transect sample for Yellow Warbler. A small number of sampling units for each species were removed from some analyses due to missing data.

Because of the sample size differences across years we evaluated whether there is any bias on the estimated occupancy results by comparing estimated occupancy for the sites that were visited each year versus an estimate of occupancy for the entire sample regardless of which points were visited. We compared occupancy estimates using the ‘predict’ function with the fitted models to predict (extrapolate) occupancy both across the entire sample (every point or transect in every year), and also for only the locations that were actually visited each year. The prediction dataset included average

values for day of year and time. The bulk of unsurveyed locations across years include 32 transects that were not surveyed in 2010-2012 (prior to rebalancing the sample in 2013), 212 transects in 2017 (after reducing field survey effort), and 235 in 2019. From the comparison of estimates we looked for consistent differences between sampled occupancy vs. extrapolated occupancy estimates for the entire sampling frame. We report analysis sample sizes and comparisons of sampled occupancy versus extrapolated occupancy means and standard errors. Note that because the method of generating the occupancy estimates varies in this analysis ('predict' function) vs. the method we described earlier (functions 'ranef' and 'bup' with bootstrapping), the occupancy estimates are slightly different.

Data Management and Access: Sierra Nevada Avian Monitoring Information Network

Additional 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 ten 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 2019, we surveyed 1585 upland point count stations (257 of which are riparian points) on 314 transects (upland and riparian combined; Table 1). We conducted two survey visits at 29% of transects for a total of 405 transect visits (2010 – 2017 range = 479 – 987). We visited 64 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, as well as very difficult access conditions due to historically high snowpack and frequent rain in May. In Sierra and Sequoia National Forest we were not able to access several sites due to falling dead trees as a result of beetle mortality.

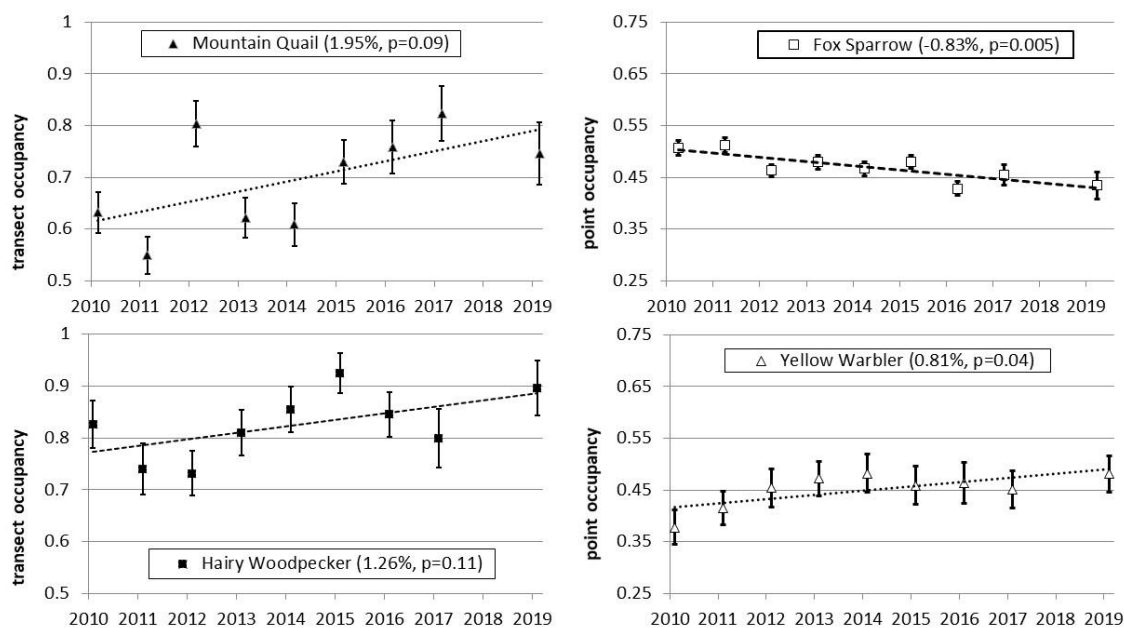
MIS Occupancy Trends

Fox Sparrows were detected at 27% of the upland point count stations we surveyed in 2019, and point scale occupancy corrected for detection probability was 0.43 (95% CI: 0.39 – 0.49). This yearly estimate is the second lowest among all years (range 0.43 – 0.51), where the highest value was in 2011 (Figure 1). The trend in Fox Sparrow occupancy averaged a statistically significant and consistent 0.83% annual decline from 2010-2019 ($P = 0.005$). Probability of detection at average covariate values in 2019 was 0.57 (95% CI: 0.43 – 0.69), which was the lowest across all years and ranged up to 0.70 in 2015.

Table 1. Survey effort by year. The full upland sample includes 474 transects, in 2017 we reduced effort by 50% and continued that sampling effort in 2019. The full riparian target sample size is 96 transects, but in 2017 we reduced the number to 68 targeting the montane riparian sites connected to wet meadows.

		2009	2010	2011	2012	2013	2014	2015	2016	2017	2019
Transects Visited	<i>upland</i>	415	464	472	462	473	474	474	474	262	250
	<i>riparian</i>	43	107	97	102	96	97	96	96	68	64
Second visits	<i>upland</i>	250	267	220	369	303	261	255	208	119	79
	<i>riparian</i>	16	65	92	57	52	55	44	58	22	12
Second visit rate	<i>upland</i>	60%	58%	47%	80%	64%	55%	54%	44%	45%	36%
	<i>riparian</i>	37%	69%	92%	56%	54%	57%	46%	60%	32%	19%

Figure 1. Multi-year occupancy estimates from 2010 – 2019 for four upland Management Indicator Species across Sierra Nevada National Forests. Error bars show ± 1 standard deviation of bootstrap samples. Linear trend per year percent increase/decrease is shown in figure legends, along with significance.



Hairy Woodpeckers were detected at 56% of the upland transects we surveyed in 2019, and transect scale occupancy was 0.90 (95% CI: 0.79 – 1.00), with a non-significant positive annual trend (1.3%, $P =$

0.11). The estimates from 2016 and 2017 appear to be lower than in 2015 (0.90) or 2019 (Figure 1). Probability of detection in 2019 at average covariate values was 0.47 (95% CI: 0.44 – 0.51) and did not vary across years.

Mountain Quail were detected at 29% of upland transect locations in 2019, and transect scale occupancy was 0.75 (95% CI: 0.63 – 0.87). There was weak evidence of a 1.95% annual increase ($P = 0.09$), but the yearly variation was large (Figure 1). Probability of detection in 2019 at average covariate values was 0.34 (95% CI: 0.20 – 0.52) and ranged from a low of 0.23 in 2016 to a high in 2011 of 0.45.

Yellow Warblers were detected at 22% of the riparian point locations we surveyed in 2019, and point scale occupancy was the near the highest across all years at 0.48 (95% CI: 0.40 – 0.57). Occupancy in other years ranged from 0.38 – 0.49 (Figure 1) with a statistically significant 0.81% per year increase ($P = 0.04$). Probability of detection was 0.51 (95% CI: 0.47 – 0.56), 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 white fir, and lower occupancy in non-forest, eastside pine, lodgepole pine, and red fir.

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

Fox Sparrow	B	SE	P(> z)
Occupancy (ψ)			
Intercept	0.15	0.09	0.10
Latitude	-0.53	0.02	0.00
Elevation	0.96	0.04	0.00
Snag Density (>10cm)	-0.09	0.04	0.02
Snag Density (>30cm)	0.18	0.05	0.00
Tree Cover	-0.21	0.04	0.00
Shrub Cover	0.24	0.03	0.00
Basal Area	-0.08	0.03	0.00
Maximum tree diameter	0.10	0.03	0.00
Precipitation	0.77	0.03	0.00
Year: 2012	-0.21	0.10	0.04
Year: 2014	-0.20	0.11	0.06

Year: 2016	-0.43	0.11	0.00
Year: 2019	-0.38	0.18	0.04
Habitat: Chaparral	0.76	0.10	0.00
Habitat: Eastside Pine	-0.15	0.07	0.03
Habitat: Lodgepole Pine	-1.83	0.13	0.00
Habitat: Red Fir	-0.88	0.09	0.00
Habitat: White Fir	0.19	0.07	0.01
Habitat: Other	-1.20	0.24	0.00

Detection Probability (ρ)			
Intercept	0.61	0.08	0.00
Tree Cover	-0.21	0.03	0.00
Shrub Cover	0.61	0.04	0.00
Time of day	0.03	0.02	0.12
Day of year	0.05	0.02	0.04
Year: 2011	0.20	0.12	0.00
Year: 2015	0.23	0.12	0.05
Year: 2016	0.22	0.13	0.08
Year: 2019	-0.35	0.19	0.06

Eight covariates were strong predictors of higher Hairy Woodpecker occupancy: higher densities of all sizes of snags, low shrub cover, low basal area, large tree diameter, southern latitudes, and they had lower occupancy in chaparral, red fir, and non-forested habitat types (Table 3). The only significant year effect was 2012, 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	1.81	0.39	0.00
Latitude	-0.45	0.14	0.00
Shrub Cover	-0.22	0.17	0.21
Basal Area	-0.29	0.15	0.06
Maximum Tree Diameter	0.31	0.14	0.02
Snag Density (>10cm)	0.24	0.21	0.26
Habitat: Chaparral	-0.90	0.42	0.03
Habitat: Red Fir	-0.54	0.32	0.09
Habitat: Other	-2.56	0.74	0.00
Year: 2012	-0.62	0.37	0.09

Detection Probability (ρ)			
Intercept	-0.10	0.06	0.10
Tree Cover	-0.09	0.04	0.05
Shrub Cover	-0.13	0.06	0.02
Snag Density (>10cm)	0.09	0.04	0.02
Time of day	-0.08	0.03	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, and with few snags of both small and large size. Significant effects of habitat types included positive for chaparral and eastside pine, but negative for lodgepole pine. The only significant year effects were 2012 and 2015 which were both positive.

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

Mountain Quail	B	SE	P(> z)
Occupancy (ψ)			
Intercept	1.14	0.34	0.00
Latitude	-2.06	0.23	0.00
Elevation	0.49	0.18	0.01
Tree Cover	-0.59	0.12	0.00
Maximum Tree Diameter	0.33	0.15	0.03
Snag Density (>10cm)	-0.18	0.12	0.12
Snag Density (>30cm)	-0.25	0.12	0.03
Habitat: Chaparral	1.48	0.45	0.00
Habitat: Eastside Pine	0.59	0.31	0.06
Habitat: Lodgepole Pine	-2.54	0.55	0.00
Year: 2012	1.48	0.51	0.00
Year: 2015	0.72	0.42	0.08
Detection Probability (ρ)			
Intercept	-0.53	0.14	0.00
Shrub Cover	0.27	0.04	0.00
Basal Area	-0.30	0.05	0.00
Snag Density (>30cm)	0.27	0.06	0.10
Time of Day	-0.23	0.04	0.00
Day of Year	-0.29	0.05	0.00
Precipitation	0.35	0.06	0.00
Year: 2011	0.33	0.20	0.10
Year: 2012	-0.54	0.18	0.00
Year: 2013	-0.35	0.19	0.07
Year: 2015	-0.43	0.19	0.02

Year: 2016	-0.71	0.20	0.00
Day of Year*precipitation	0.14	0.05	0.00

There were eight significant predictors of Yellow Warbler occupancy: northern latitude, lower elevations, low tree cover, low shrub height, and high cover of grass, leaf litter, and forbs in the ground layer, plus several years had significant positive effects on occupancy (Table 5).

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.

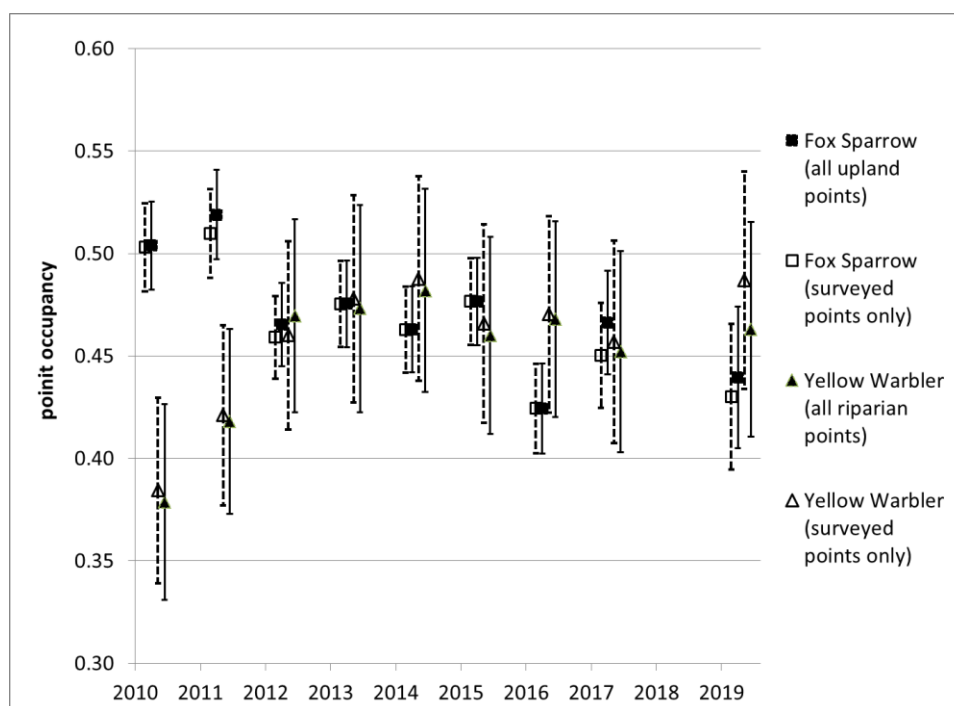
Yellow Warbler	β	SE	$P(> z)$
Occupancy (ψ)			
Intercept	-0.76	0.26	0.00
Latitude	1.06	0.10	0.00
Elevation	-1.82	0.13	0.00
Tree Cover	-0.58	0.10	0.00
Shrub Height	-0.99	0.13	0.00
Grass Cover	0.50	0.15	0.00
Leaf litter Cover	0.55	0.13	0.00
Forbs Cover	0.53	0.13	0.00
Year: 2012	0.64	0.33	0.05
Year: 2013	0.67	0.35	0.05
Year: 2014	0.73	0.34	0.03
Year: 2015	0.58	0.34	0.09
Year: 2016	0.63	0.33	0.06
Year: 2019	0.60	0.36	0.10
Detection Probability (ρ)			
Intercept	0.04	0.09	0.62
Tree Cover	0.38	0.10	0.00
Shrub Cover	0.47	0.09	0.00
Shrub Height	1.26	0.12	0.00
Basal Area	-0.51	0.11	0.00
Day of Year	-0.17	0.07	0.01
Time of day	0.24	0.06	0.00

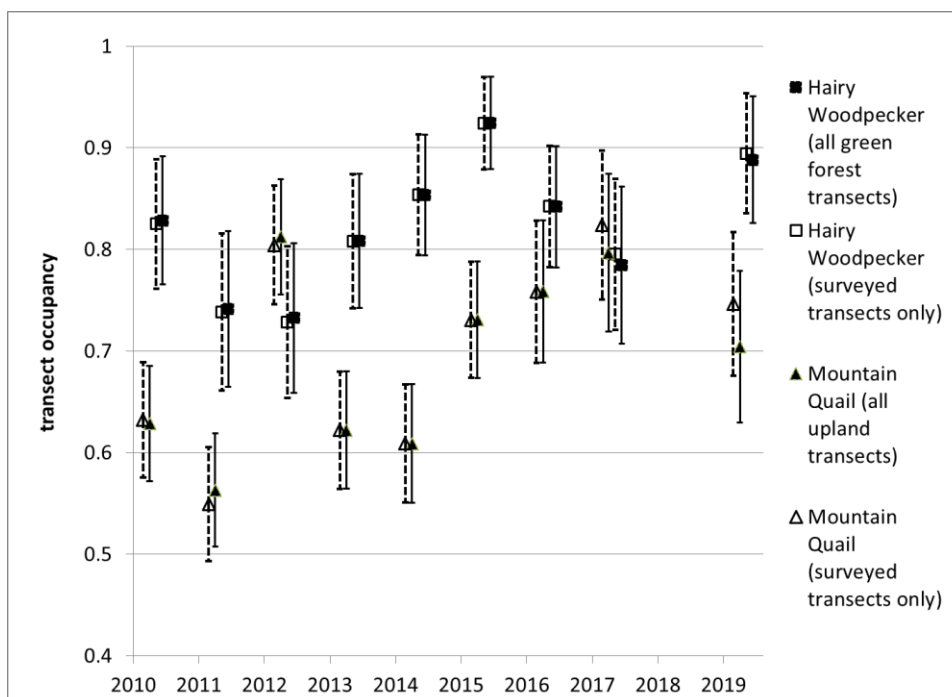
Sampled vs. Extrapolated Occupancy

Comparing the occupancy estimates for each species between using only the surveyed locations from each year versus extrapolating the average occupancy estimate across all locations in the sampling frame shows that the estimates are very similar overall (Figure 2). The only apparent discrepancies occur during years when a larger proportion of the sample was not surveyed. During the years of 2013-2016 nearly all locations in the sample were surveyed for all species, but in the other years the surveyed sample size varied. For Fox Sparrow on all upland point count locations ($n=2353$), the lowest proportions

of surveyed sample occurred in 2010-2012 ($n=2026, 2173, 2202$ respectively), and in 2017 and 2019 ($n=1297$ and 1184 respectively). The largest difference between average occupancy on surveyed locations vs. all locations occurred in 2017, but even that difference was very small (0.458 vs 0.472). Yellow Warbler occupancy estimates were almost identical between the surveyed and complete samples despite a much smaller sample size overall ($n=272$ points in total). But that was to be expected given that most of the surveyed locations were visited every year after we removed the non-meadow sites. The largest difference between occupancy estimates was in 2019 ($n=257$), but that difference was very small (0.487 vs. 0.463). Hairy Woodpecker occupancy estimates were remarkably similar across all years despite the sample of green forest transects ($n=402$) being much lower in 2017 ($n=223$) and 2019 ($n=213$). Mountain Quail estimates on upland transects ($n=474$) were similar through 2016, but differed in 2017 ($n=262$) and 2019 ($n=239$) with occupancy on surveyed transects higher compared to all upland transects.

Figure 2. MIS occupancy estimates in each individual year generated from surveyed locations only vs. extrapolated to all locations in the sampling frame. Error bars show ± 1 standard error of occupancy estimates.





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, 2016, 2018) we have detected sizeable year to year variation in occupancy, including significant or marginally significant trends for all four species that may indicate a consistent directional change across the ten years that span this monitoring project. An average annual decline in Fox Sparrow occupancy of 0.83% appears to be highly consistent and thus could be an ecologically significant pattern. Fox Sparrow is the only species of the four MIS that appears to be declining. Yellow Warbler has increased 0.81% per year and this trend is statistically significant, while the distributions of Hairy Woodpecker and Mountain Quail are not statistically significant even though the magnitude of their increases are larger (1.95% for Mountain Quail, and 1.26% for Hairy Woodpecker). The year-to-year variation for all three of these species implies that their patterns could be partly driven by some more complex set of conditions rather than a constant linear mechanism. The 2019 Hairy Woodpecker occupancy estimate is the second highest we have recorded, and that follows a relatively low estimate just two years earlier in 2017. The recent high

values in Mountain Quail occupancy (since 2015) lead to a strong positive but marginally 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 to possibly increasing from a short-term (<10 years) perspective, while Fox Sparrow appears to be declining. 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 9 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 2019 saw a historically high winter snowpack, as well as wet weather conditions early in the season (May). 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, we found significant effects of drought, warming, and conifer mortality on many species (Roberts et al. 2019).

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 (this is the focus of new manuscript in prep). 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 in 2019 at 43% 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 two northern most forests and the eastside Inyo National Forests (Roberts et al. 2013). We detected a statistically significant and consistent decline in Fox Sparrow occupancy from 2010 to 2019. Given the pattern in occupancy displayed by this species, drought does not appear to be a cause of the decline. However, a number of management actions can alter habitat suitability for Fox Sparrow. These include fire suppression which results in a slow succession of more open forest into closed-canopy forest and the loss of gaps and openings that once supported shrub patches. More immediate are the short term effects of fire burning forested habitat, and the effects of management activities that may reduce shrub density and extent. Though fire has a short term negative effect on species occurrence (presumably through the immediate removal of shrub and ground cover), Fox Sparrows reach some of their greatest densities in fire-created chaparral/shrublands a decade or more following fire. 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 shrublands 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. 2019). 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 and a more nuanced analysis of changing vegetation conditions and management activities across our sample could inform whether efforts are needed to improve the habitat conditions for this shrub associated indicator species in Sierra Nevada national forests.

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 consistent trend in distribution over time, although 2019 occupancy was the second highest we have recorded. The pattern of occupancy across time appears to be somewhat cyclical with up to three years of decline (e.g. 2010-2012, and 2015-2017) followed by increases (e.g. 2013-2015, and 2019). 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 population densities. Detection probability for this species is also relatively low, even with playback, such that field survey counts considerably underestimate the species' true occupancy.

Hairy Woodpeckers 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 moderate to high severity burned forest (Burnett et al. 2012), as well as low severity fire and management that reduces understory cover (unpublished data). Hairy Woodpecker has 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 restoration (Campos & Burnett 2014) and shaded fuel breaks. Our evaluation of their occupancy here excludes areas that have burned since 2000, thus it is an underestimate of the species occupancy across all habitats in the region since they likely occupy a high proportion of those recently burned forest patches.

Mountain Quail

Mountain Quail occupancy was higher in 2012, 2015, 2016, 2017, and 2019 than in other years, and a marginally significant positive linear trend in population distribution is apparent. High inter-annual variation in detectability may be related to nesting phenology due to weather patterns in late winter and early spring causing Mountain Quail to reduce vocalization rates during active incubation and chick brooding 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).

As they are the indicator for early and mid-seral conifer forest, we have found that this species is most abundant in shrub-dominated habitats. But, unlike Fox Sparrow, Mountain Quail also 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. Further, fuels reduction and restoration treatments in mature forest that promote openings and gaps, while also facilitating growth of large diameter trees, could benefit this species.

Yellow Warbler

As of 2019, Yellow Warblers occur at over 43% of our meadow-associated riparian point count stations and they appear to be increasing. The fact that they do not occur at up to 55% of our meadow sampling stations suggests that their distribution may be limited by a variety of factors such as elevation, tree encroachment into riparian meadows, and shrub structure and composition. Occupancy was higher at low elevation locations with low shrub height, low tree cover, and high ground cover including grass/sedge, forbs, 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, will likely 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 ten year data set represents a significant investment on the part of the USFS and Point Blue, and its value will 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 used these data in an analysis of the avian community in post-fire habitat (Taillie et al. 2018). We recently published a manuscript that evaluated the response of the avian community to the massive recent drought and tree mortality event in the southern Sierra (Roberts et al. 2019). Currently we are preparing a manuscript comparing the long-term effects of fire and silvicultural treatments on the bird community. 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 trends assessment, evaluation of species responses to climate changes, and assessments of responses of bird species to disturbances including fire and management. Fortunately, statistical confidence in MIS occupancy estimates was minimally affected in 2017 and 2019 as a function of smaller sample size. In addition we found that the occupancy estimates are very similar between the half and full samples during 2017 and 2019, indicating that our original methods employing GRTS to subset our sample while preserving the representation of the full sample was successful. However, by only sampling half the sample every other year we are missing opportunities to record post-disturbance changes for some species where fires and other disturbances occur on our un-surveyed sampling locations, and we also lose resolution of bird distribution responses over time after disturbances. 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 in previous reports (Roberts et al. 2015, 2016, 2017, 2018) 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 and 2019 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, but changed the criteria to define fire influence slightly in comparison to past reports in order to increase the sampled area. These transects are defined as areas that had not burned at moderate or high severity from 2000-2019 (previously 1996-2017) and were more than 2 km from recent moderate or high severity fire patches from 2006-2019 (previously 2004-2017). This process removed 127 of the 474 upland transects, including 14 transects that have burned since our last field season in 2017, leaving a sample size of 347 for this analysis (previously $n = 362$ 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.

We assessed changes in the Black-backed Woodpecker population distribution from 2011 to 2019 using the 'occu' occupancy function from the package 'unmarked' (Fiske and Chandler 2011, Kery and Chandler 2012) in program R version 3.3 (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, and also excluded 2010 as we had not yet implemented playback surveys for Black-backed Woodpecker. In previous reports 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), to evaluate Black-backed Woodpecker patterns of occurrence. But with that method it is difficult to include covariates that vary across years, e.g. shrub cover or snag count changes would have to be constant across all years in the data. As of 2019 we have data that span 10 years and over this amount of time many of our vegetation measurements have changed significantly from early in the sampling period to the current surveys. So for this report we have altered the occupancy method to the same structure use for the other MIS (described in Methods pages 5-8 earlier in this report). One important difference is that we consider the Black-backed Woodpecker playback surveys as separate survey events from the combined passive point counts and the Hairy Woodpecker and Mountain Quail playback surveys as we had done in previous analyses. This means that in each year there are four total survey events, and for each of the $n = 347$ green forest transects there were a maximum of 8 years of surveys. Since not all 347 transects were visited each year the number of transect/year sampling units varies: 323 in 2011, 327 in 2010, 347 in 2013-2015, 346 in 2016, 182 in 2017, and 173 in 2019. We included most of the same occupancy covariates as previous reports, including latitude, elevation, slope, southness, tree cover, shrub cover, basal area, maximum tree diameter, two snag size counts ($>10\text{cm}$ and $>30\text{cm}$), habitat type, and year. The detection probability portion of the model included shrub cover, tree cover, basal area, snag counts, time of day, day of year, year, and a binary value for survey type (passive plus Hairy Woodpecker and Mountain Quail playback surveys = 0, Black-backed Woodpecker playback survey = 1).

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]. Model selection proceeded using a standard AIC stepwise removal approach where each variable with the lowest probability was removed until AIC did not improve. Means and variances of location-level occupancy were estimated using 500 parametric bootstraps and an empirical Bayes method with the functions 'ranef' and 'bup' (Fiske and Chandler 2011). Reported values are means, standard deviation of bootstrap estimates (for error bars in figures), and 2.5% and 97.5% quantiles (as an estimate of 95% confidence interval) of the bootstrap estimates.

Because of the sample size differences across years we evaluated whether there is any bias on the estimated occupancy results by comparing estimated occupancy for the sites that were visited each year versus an estimate of occupancy for the entire sample regardless of which points were visited. We

compared occupancy estimates using the ‘predict’ function with the fitted model to predict (extrapolate) occupancy both across the entire sample (every point or transect in every year), and also for only the locations that were actually visited each year. The prediction dataset included average values for day of year and time. The bulk of unsurveyed locations across years include 20 and 24 transects that were not surveyed in 2011 and 2012 (prior to rebalancing the sample in 2013), 165 transects in 2017 (after reducing field survey effort), and 173 in 2019. From the comparison of estimates we looked for consistent differences between sampled occupancy vs. extrapolated occupancy estimates for the entire sampling frame. We report analysis sample sizes and comparisons of sampled occupancy versus extrapolated occupancy means and standard errors. Note that because the method of generating the occupancy estimates varies in this analysis (‘predict’ function) vs. the method we described earlier (functions ‘ranef’ and ‘bup’ with bootstrapping), the occupancy estimates are slightly different.

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	2019
Modoc	6 (37)	6 (37)	4 (39)	4 (39)	8 (39)	5 (39)	2 (14)	2 (14)
Lassen	8 (57)	11 (55)	10 (57)	7 (57)	9 (57)	7 (57)	5 (27)	6 (25)
Plumas	0 (38)	2 (38)	0 (38)	2 (38)	0 (38)	1 (38)	0 (20)	1 (18)
Tahoe	2 (42)	3 (46)	4 (46)	2 (46)	4 (46)	3 (45)	1 (20)	1 (18)
LTBMU	0 (2)	0 (2)	0 (2)	0 (2)	0 (2)	0 (2)	0 (0)	0 (0)
Eldorado	1 (38)	2 (38)	4 (41)	4 (41)	3 (41)	3 (41)	4 (19)	3 (18)
Stanislaus	0 (18)	4 (22)	4 (24)	0 (24)	4 (24)	0 (24)	0 (10)	2 (14)
Inyo	5 (18)	2 (18)	6 (20)	6 (20)	4 (20)	8 (20)	3 (14)	5 (14)
Sierra	5 (49)	4 (47)	6 (53)	5 (53)	5 (53)	13 (53)	4 (39)	6 (37)
Sequoia	0 (24)	0 (24)	0 (27)	0 (27)	0 (27)	1 (27)	0 (19)	0 (15)
Total	27 (323)	34 (327)	38 (347)	30 (347)	37 (347)	41 (346)	19 (182)	26 (173)

Results and Discussion

Black-backed Woodpeckers have been detected on at least one green forest transect in all forests except for the Lake Tahoe Basin Management Unit, but we only have 2 transects in the basin (Table A1) and they have been detected there on burned transects. In addition we only have a single detection within Sequoia National forest. However, other researchers have detected Black-backed Woodpeckers on the Sequoia in burned forest (Siegel et al. 2015a). At least one Black-backed Woodpecker has been detected at a total of 133 of the 347 green forest transects (38%) from 2011 to 2019.

The lowest AIC model following the stepwise removal process included eight occupancy and three detection covariates (Table A2). Occupancy is positively associated with low slopes, high elevations, northern latitudes, high tree cover, high density of small snags and low density of large snags, Eastside Pine and White Fir habitat types, and negatively associated with south-facing aspects. Detection probability covariates included a positive effect of >10cm snag density, low shrub cover, and day of year (later in the season).

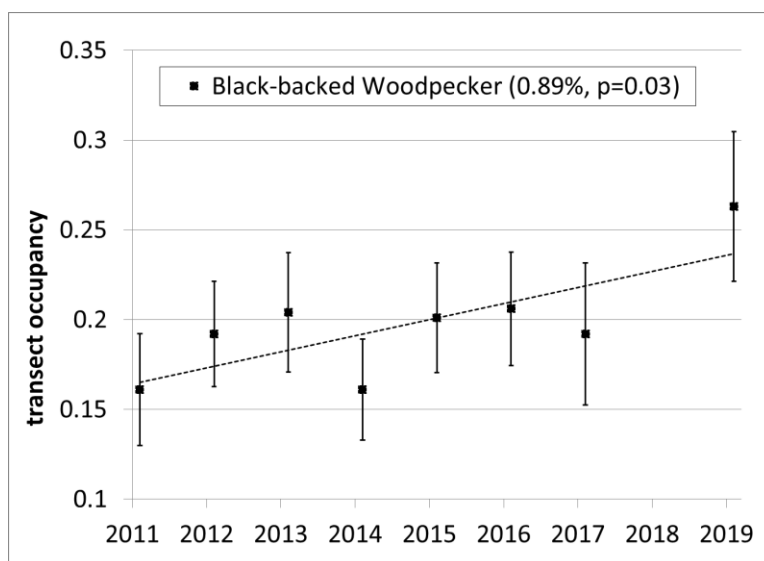
Table A2. Final occupancy model coefficients chosen via AIC-based stepwise variable removal.

Black-backed Woodpecker	B	SE	P(> z)
Occupancy (ψ)			
Intercept	-2.71	0.34	0.00
Elevation	1.29	0.17	0.00
Latitude	0.22	0.11	0.04
Slope	-0.80	0.12	0.00
Southness	-0.24	0.09	0.01
Tree cover	0.12	0.10	0.22
Snag Density (>10cm)	0.22	0.16	0.18
Snag Density (>30cm)	-0.27	0.16	0.08
Habitat: Eastside Pine	0.59	0.30	0.05
Habitat: White Fir	0.54	0.31	0.08
Year: 2019	1.15	0.41	0.00
Detection Probability (ρ)			
Intercept	-1.25	0.13	0.00
Shrub cover	-0.45	0.10	0.00
Snag density (>10cm)	0.47	0.14	0.00
Day of year	0.18	0.14	0.04

Estimated occupancy ranged from 0.16 in 2011 and 2014 to 0.26 in 2019 with error bars largely overlapping for all years prior to 2019 (Figure A1). The occupancy estimates on average across years are very similar to our previous analyses, though there is more year-to-year variation now that we have

moved to the new model structure and yearly occupancy is assessed independently from other years (i.e. transect colonization and extinction are not explicitly incorporated into the model). The pattern across years shows a significant increasing trend of 0.89% per year ($P = 0.03$), but this appears to be largely driven by the high value in 2019. At this point it is unclear whether the 2019 occupancy value represents an outlier, a sampling artifact, or a real increase in the Black-backed Woodpecker population distribution. The relatively steady pattern from 2011 to 2017 aligns with our previous results (Roberts and Burnett 2016, 2017, 2018), but contradicts the potentially decreasing trend we reported following the 2014 field season, which was also driven by one particularly low value (Roberts et al. 2015).

Figure A1: Annual occupancy estimates for Black-backed Woodpeckers in green forest in each individual year generated from surveyed locations only vs. extrapolated to all locations in the sampling frame. Error bars show ± 1 standard error of occupancy estimates.

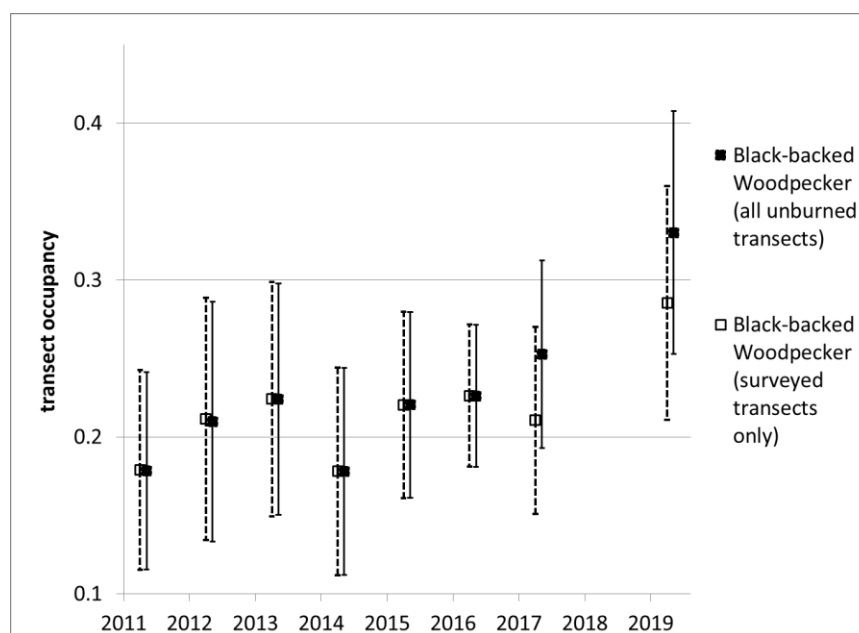


Even though the 2019 estimate of 0.26 is an abnormally high value compared to other years, the bootstrap results show that this value justified by the model and data (bootstrap sample 2.5% quantile = 0.19, and 97.5% quantile = 0.36). One possible reason for this result could be that we happened (randomly) to survey a larger proportion of high quality Black-backed Woodpecker habitats in 2019 compared to other years (but see Figure A2 and later discussion). They were detected at 26/173 transects in 2019, naïve occupancy = 0.15, which is higher than all other years which range from 0.08 to 0.12 (Table A1). As in previous reports, we suggest that it is unclear whether this result can be interpreted as a significantly positive trend in Black-backed Woodpecker occupancy in Sierra Nevada region green forest, or as an indication that the population distribution is stable. It is also important to note that since the transects included samples have changed across years as some locations become influenced by recent fires, or other locations transition into the green forest sample due to time passing

since older fires, the occupancy results from past reports are not directly comparable to the results shown in this report.

Comparing estimated occupancy of the full unburned sample vs. the surveyed locations only shows that when half the sample was not surveyed in 2017 and 2019 the occupancy estimates differed noticeably (Figure A2). But the extrapolation of predicted occupancy across the entire sample including all unsurveyed locations only increases the estimates even more. Thus we can conclude that the very high occupancy estimate in 2019 is not likely to be an artifact of inadvertently surveying high quality locations, but rather is an unbiased estimate of Black-backed Woodpecker occupancy across our study area in unburned forest locations.

Figure A2: Annual occupancy estimates for Black-backed Woodpeckers in green forest. Vertical lines bounding each point indicate 1 standard deviation of difference between model estimated occupancy and 500 bootstrap estimates.



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 (small snags over 10cm and less than 30cm in diameter). In this analysis we found one notable difference. That difference is that lodgepole pine and red fir habitat types were not significant, but rather eastside and Jeffrey pine, and white fir habitat types were positive influences on occupancy. We suspect that the elevation covariate is accounting for Black-backed Woodpecker occupancy in lodgepole and red fir forest habitats, while at the lower end of their elevation distribution they may prefer eastside/Jeffrey pine over other habitats that are in the same elevation range (e.g. Sierra mixed conifer,

Douglas fir, and ponderosa pine). Another interesting result is that snags >10cm in diameter were a positive influence on occupancy, while snags >30cm were a negative influence.

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 marginally 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.

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