Point Blue Report
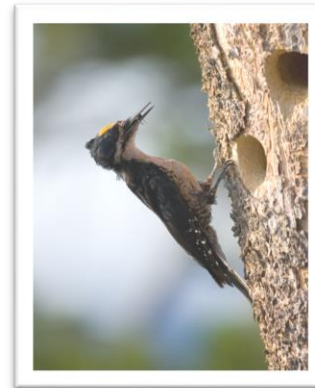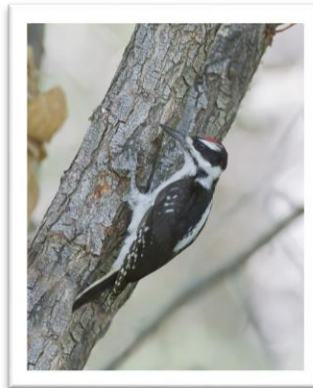
Sierra Nevada National Forests Avian Management Indicator Species

2015 Annual Report

June 10, 2016

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Sierra Nevada National Forests Avian Management Indicator Species Project

Point Blue Conservation Science, March 2016

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SUMMARY

2015 marked the seventh year of monitoring four avian Management Indicator Species (MIS) across 10 National Forest units in the Sierra Nevada planning area. In 2015, we used multi-species point counts to sample 474 transects in upland habitat for Fox Sparrow, Hairy Woodpecker, and Mountain Quail. We surveyed an additional 96 transects in riparian habitats for Yellow Warbler.

We investigated MIS distribution trends using a dynamic occupancy model incorporating a large set of covariates on occupancy, detection probability, and probability of survey locations being colonized or abandoned. Occupancy trends indicate that Fox Sparrow and Mountain Quail distributions are stable across 2010-2015, while Hairy Woodpecker and Yellow Warblers are increasing slowly but significantly (~1% per year). The addition of a large set of covariates on the dynamic model parameters has reduced uncertainty in occupancy rates and increased the confidence in trend estimates. Prior to the addition of this sixth year of survey data we have had difficulty with model fit with such a large set of covariates and such a complex model.

The value of bioregional monitoring programs such as this ongoing Sierra Nevada project goes far 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), and identified baseline reference locations for post-fire analyses in Lassen, Eldorado, and Stanislaus National Forests (Fogg et al. 2015). We are currently investigating the long-term effects of fire and silvicultural management on the avian community across our study area. We plan to continue using the Sierra Nevada bioregional monitoring dataset to help inform forest management across this ecologically important mountain range.

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 land.

In this report we describe the field efforts and results updated through the 2015 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 2004a). 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 sites retains the overall spatial balance, allowing for easy adjustment to the number of sites 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 different National Forests. 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 goal 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.

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 management actions. There are approximately 30-90 transects per forest (150-450 point count locations), and most power analyses recommend at least 60 survey sites for reasonably precise and unbiased occupancy analyses given typical conditions (e.g. McKann et al. 2013).

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 sites (points or transects) occupied by

correcting raw counts for probability of detection (MacKenzie et al. 2006). Uncorrected counts can be misleading 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 used multiple-season occupancy models to assess changes in MIS population distribution from 2010 to 2015 using the 'colext' 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 transects 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 included a set of covariates on occupancy including CWHR habitat type (Mayer and Laudenslayer 1988, USDA Forest Service 2004b) simplified 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". Other remotely sensed covariates included latitude, elevation corrected for latitude, yearly precipitation (PRISM Climate Group 2004), and solar radiation index (SRI), which is a linear representation of aspect (Keating et al. 2007). Tree cover, shrub cover, counts of snags greater than 10cm in diameter, and total basal area covariates were estimated using habitat assessment surveys (see Roberts et al. 2013). The riparian data include a slightly different set of covariates (latitude, elevation, tree cover, shrub cover, shrub height, basal area, and ground percent cover variables: forbs, grass, litter, barren). We chose final occupancy models for each species by iteratively removing one occupancy covariate with the lowest significance until AIC did not improve. Model selection results are not shown. See Roberts et al. (2013) for an analysis of MIS habitat and

topographic associations. Covariates on detection, colonization, and extinction included the same set of variables as occupancy and were iteratively chosen using AIC in the same manner as occupancy. During selection of covariates for occupancy the other three parameters were held constant (no covariates), then detection covariates were added and selected, followed by colonization, and finally extinction covariates. Standard errors for all model parameters were estimated using 100 non-parametric bootstraps.

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 removed all transects with at least one point that was located within 50m of a patch of medium to high severity fire that occurred in 1995 or later to minimize detections of birds within burned 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 2015 green forest sample includes 414 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 2394 points on 474 transects. The riparian

sample includes 374 points on 94 transects. A small number of points in each data set were removed due to missing covariate data.

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 the playback surveys are included for transect-scale occupancy analyses, in previous annual reports we had used only passive point count detections.

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 six years of this project, we have amassed over 380,000 individual bird records of 202 species at approximately 3000 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 2015, we surveyed 2750 point count stations on 570 transects (upland and riparian combined; Table 1). We conducted two survey visits at 52% of transects for a total of 869 transect visits (compared to 884 in 2014, 924 in 2013, 987 in 2012, 876 in 2011, and 890 in 2010). We conducted two visits at all but four transects where only single visits had been performed in 2014.

Table 1. Survey effort by year. The target upland sample includes 474 transects. 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
Transects Visited	*Upland*	415	464	472	462	473	474	474
	riparian	43	94	96	100	96	97	96
Second visits	*upland*	250	267	220	369	303	261	255
	riparian	16	65	88	56	52	55	44
Second visit rate	*upland*	60%	58%	47%	80%	64%	55%	54%
	riparian	37%	69%	92%	56%	54%	57%	46%

MIS Occupancy Trends

Fox Sparrows were detected at 38% (up from 34% in 2014) of upland point count stations in 2015, and point scale occupancy corrected for detection probability was 0.47 (95% CI: 0.45 – 0.50). From 2010 – 2015 occupancy ranged from 0.47 – 0.49, highest in 2010 and lowest in 2014 (Figure 1). Occupancy has declined at a rate of -0.31% per year from 2010 – 2015 ($P = 0.05$). Probability of detection in 2015 was 0.65 (95% CI: 0.56 – 0.72) and ranged from 0.60 – 0.72 across years.

Hairy Woodpeckers were detected at 58% (up from 54% in 2014) of upland transects in 2015, and transect scale occupancy was 0.92 (95% CI: 0.85 – 1.00), increasing steadily at a rate of 1.4% ($P < 0.01$) (Figure 1). Prior to including covariates and survey data from 2015 our models showed a steady 3% per year decline from 2010-2013 (Roberts et al. 2015), but that declining trend is no longer supported presumably due to additional survey data and covariate information. Probability of detection was higher at 0.43 (95% CI: 0.30 – 0.57) in 2015 than in previous years which ranged from 0.33 – 0.42.

Mountain Quail were detected at 29% (compared to 28% in 2014) of upland transect locations in 2015, and transect scale occupancy was 0.65 (95% CI: 0.58 – 0.71) in 2015. Occupancy was steady across years, ranging from 0.63 – 0.65 across all five years (Figure 1), with no trend. Probability of detection was 0.36 (95% CI: 0.32– 0.40) and did not vary across years (the AIC variable selection process removed year as a covariate of detection).

Yellow Warblers were detected at 22% (compared to 23% in 2014) of riparian point locations in 2015, and point scale occupancy was 0.35 (95% CI: 0.29 – 0.41). Occupancy in other years ranged from 0.31 – 0.34 (Figure 1) and showed a significant positive trend

of 0.8% per year ($P = 0.02$). Probability of detection was 0.61 (95% CI: 0.57 – 0.65) with no variation across years.

Figure 1. Multi-year occupancy estimates in 2010 – 2015 for the MIS. Error bars show +/-1 standard error derived from 100 non-parametric bootstraps of the multi-season occupancy model.
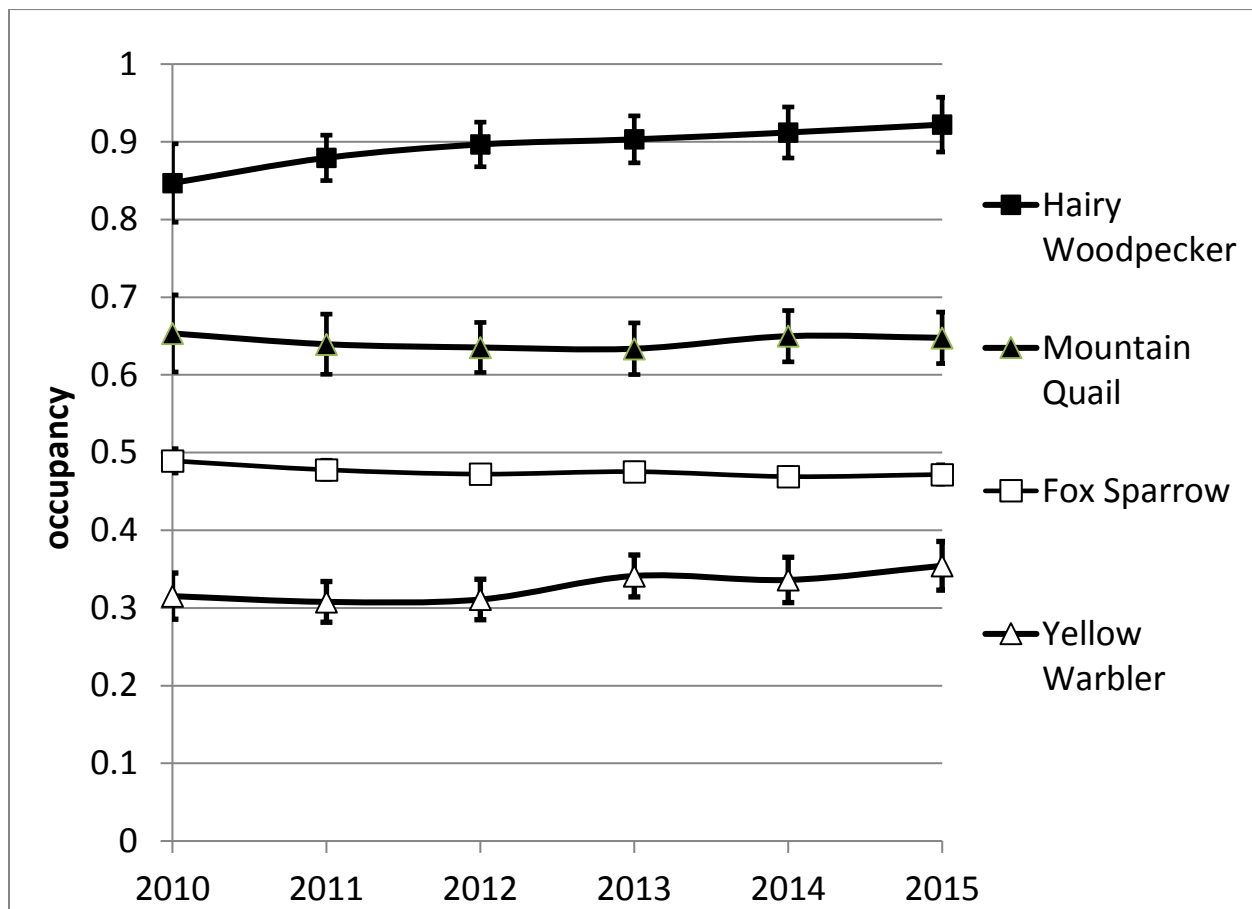
Covariates of Occupancy

Evaluating covariate associations with MIS occupancy, colonization, and extinction can provide useful information on the influence of various habitat parameters on these species distribution patterns. Seven covariates significantly influenced Fox Sparrow occupancy (Table 2). Within our sampling frame occupancy increased at more northerly

latitudes, at higher elevations, in areas with lower tree and higher shrub cover, and areas with higher precipitation. Across habitat types they had lower occupancy in lodgepole pine and red fir. Colonization was higher at lower latitudes, higher elevations, in areas with less tree cover, and greater precipitation. Extinction covariates were similar to those for occupancy, but the coefficients tended to have the opposite sign indicating that extinction was more likely to occur at locations with less suitable habitat.

Table 2. Standardized coefficient estimates, unconditional standard errors, and P-value derived from the lowest AIC dynamic occupancy model for Fox Sparrow. Non-signficant (P>0.1) habitat type and year effects have been removed.

Fox Sparrow	β	SE	P(>|z|)
Occupancy (ψ)			
Intercept	0.23	0.11	0.04
Latitude	-0.43	0.06	0.00
Elevation	0.84	0.07	0.00
Tree Cover	-0.24	0.06	0.00
Shrub Cover	0.41	0.06	0.00
Precipitation	0.77	0.07	0.00
Habitat: Lodgepole Pine	-2.09	0.32	0.00
Habitat: Red Fir	-1.11	0.21	0.00
Colonization (γ)			
Intercept	-2.67	0.08	0.00
Latitude	-0.18	0.08	0.02
Elevation	0.33	0.06	0.00
Tree Cover	-0.28	0.08	0.00
Precipitation	0.55	0.07	0.00
Extinction (ε)			
Intercept	-2.55	0.16	0.00
Latitude	0.25	0.10	0.01
Elevation	-1.10	0.17	0.00
Shrub Cover	-0.37	0.09	0.00
Precipitation	-0.30	0.10	0.00
Habitat: Chaparral	-0.73	0.38	0.05
Habitat: Lodgepole Pine	1.41	0.54	0.01
Habitat: Red Fir	1.24	0.28	0.00

Detection Probability (ρ)			
Intercept	0.60	0.08	0.00
Latitude	-0.42	0.03	0.00
Tree Cover	-0.29	0.03	0.00
Shrub Cover	0.36	0.03	0.00
Precipitation	0.25	0.04	0.00
Habitat: Chaparral	0.53	0.13	0.00
Habitat: Lodgepole Pine	-0.14	0.21	0.50
Habitat: Unforested	-0.97	0.25	<0.01
Habitat: White Fir	0.21	0.08	0.01
Year: 2011	0.32	0.09	<0.01
Year: 2013	-0.20	0.09	0.03

Only two covariates were strong predictors of higher Hairy Woodpecker occupancy: high elevations and southern latitudes (Table 3). The model also included weak effects of tree cover (negative) and basal area (positive). Areas with lower tree cover, greater live tree basal area, and high DBH were associated with a marginally significant higher colonization rate. Extinction probability was higher in areas with lower tree cover, fewer snags, and lower elevations.

Table 3. Standardized coefficient estimates, unconditional standard errors, and P-value derived from the lowest AIC dynamic occupancy model for Hairy Woodpecker. Non-signficant (P>0.1) habitat type and year effects have been removed.

Hairy Woodpecker	β	SE	P(>|z|)
Occupancy (ψ)			
Intercept	2.15	0.50	0.00
Latitude	-0.53	0.27	0.05
Elevation	0.79	0.31	0.01
Tree Cover	-0.43	0.31	0.16
Basal Area	0.68	0.48	0.16
Colonization (γ)			
Intercept	0.45	0.52	0.90
Tree Cover	-0.69	0.43	0.11
Basal Area	1.09	1.75	0.08
DBH	0.45	0.31	0.15

Extinction (ε)			
Intercept	-5.43	1.44	0.00
Elevation	-0.76	0.40	0.06
Tree Cover	-1.87	0.86	0.03
Snags	-1.49	0.79	0.06
Detection Probability (ρ)			
Intercept	-0.41	0.14	<0.01
Elevation	-0.15	0.05	0.01
Precipitation	-0.14	0.05	<0.01
Habitat: Eastside Pine	0.29	0.12	<0.01
Habitat: Red Fir	0.16	0.14	0.05
Habitat: White Fir	0.22	0.12	0.02
Year: 2012	-0.29	0.15	0.02

Five covariates explained variation in Mountain Quail occupancy (Table 4). Mountain Quail were more likely to occur further south, at lower elevations, with higher precipitation, with high shrub cover and low tree basal area. They showed no association with forest type. Colonization was higher at lower latitudes. Extinction was higher at lower latitudes, higher elevations, and sites with less shrub cover, less precipitation, and lower solar radiation (northerly aspects).

Table 4. Standardized coefficient estimates, unconditional standard errors, and P-value derived from the lowest AIC dynamic occupancy model for Mountain Quail. Non-signficant (P>0.1) habitat type effects have been removed.

Mountain Quail	B	SE	P(>|z|)
Occupancy (ψ)			
Intercept	1.45	0.37	0.00
Latitude	-1.35	0.34	0.00
Elevation	-0.77	0.32	0.01
Shrub Cover	1.27	0.40	0.00
Basal Area	-1.20	0.36	0.00
Precipitation	0.92	0.28	0.00
Colonization (γ)			
Intercept	-1.93	0.21	0.00
Latitude	-0.52	0.16	0.00

Extinction (ε)			
Intercept	-3.67	0.62	0.00
Latitude	-0.80	0.38	0.04
Elevation	2.10	0.46	0.00
Shrub Cover	-0.83	0.26	0.00
Precipitation	-0.83	0.27	0.00
Solar Radiation Index	-0.49	0.22	0.03
Detection Probability (ρ)			
Intercept	-0.56	0.09	0.00
Latitude	-0.38	0.07	0.00
Elevation	0.32	0.07	0.00
Tree Cover	-0.12	0.06	0.06
Basal Area	-0.22	0.07	0.00
Habitat: Eastside Pine	-0.46	0.16	0.00
Habitat: Lodgepole Pine	-0.83	0.30	0.01
Habitat: Unforested	-1.55	0.58	0.01
Habitat: White Fir	0.49	0.17	0.00

There were four predictors of Yellow Warbler occupancy: southern latitudes, lower elevations, higher shrub height, and low basal area (Table 5). Colonization was higher further south, in areas with higher tree cover and shrub cover, and low basal area. Extinction increased further north and at higher elevations.

Table 5. Standardized coefficient estimates, unconditional standard errors, and P-value derived from the lowest AIC dynamic occupancy model for Yellow Warbler.

Yellow Warbler	β	SE	P(>|z|)
Occupancy (ψ)			
Intercept	-0.90	0.15	0.00
Latitude	-0.32	0.19	0.09
Elevation	-0.50	0.17	0.00
Shrub Height	0.25	0.15	0.10
Basal Area	-0.69	0.28	0.02
Colonization (γ)			
Intercept	-3.23	0.23	0.00
Latitude	-0.39	0.20	0.06
Tree Cover	0.45	0.20	0.03
Shrub Cover	0.62	0.16	0.00

Basal Area	-0.73	0.42	0.08
Extinction (ε)			
Intercept	-3.31	0.63	0.00
Latitude	3.03	1.18	0.01
Elevation	1.99	0.60	0.00
Detection Probability (ρ)			
Intercept	0.45	0.09	0.00
Latitude	0.68	0.09	0.00
Shrub Cover	0.21	0.09	0.02
Basal Area	-0.23	0.13	0.08
Grass Cover	0.82	0.13	0.00
Litter Cover	0.63	0.11	0.00
Bare Ground Cover	0.30	0.12	0.01

DISCUSSION

The Sierra Nevada bioregional monitoring program tracks trends in avian MIS 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 management decisions at multiple scales from the entire Sierra Nevada region down to individual forests, and help recognize large scale changes in habitat conditions related to overriding factors such as climate change.

This project is an example of a large scale ecological monitoring effort that provides information on a broad suite of species using a single standardized methodology. The large size of this sample makes it useful for assessing associations with fine-scale habitat features, management or other disturbances, opportunistic before-after-control studies (e.g. when wildfires occur on our field sites), space-for-time analyses, and evaluating occupancy or abundance shifts on elevational gradients and other smaller geographic areas. If continued long-term, this monitoring program can provide a wealth of information to help inform forest management and conserve biodiversity in the face of

accelerating threats from climate change and increased demand on the Sierra Nevada's natural resources.

A sixth year of data appears to have met a critical mass of information to inform this data-hungry analysis, allowing the models to fully evaluate these dynamic distributions and include a full set of covariates. The contrast to our previous reports (Roberts et al. 2014, 2015) is especially apparent for Mountain Quail and Hairy Woodpecker, both of which had previously appeared to show strong year to year variation in occupancy (trends were not significant), now appear to be highly stable (Mountain Quail) or consistently increasing (Hairy Woodpecker). The significant decline in Fox Sparrow occupancy was again apparent, though the rate is slower than it was in our 2014 report (-0.3% vs -1.1%). Yellow Warblers again show a positive trend in occupancy, and now the trend is significant in contrast to our previous report.

MIS Summaries

Given the climate variability over the 6 years of monitoring, it is remarkable how steady the trends in occupancy are for these four species. The Sierra Nevada endured disparate weather conditions from above average snowfall in 2010 and 2011 to severe drought in 2013 through 2015. It is important to recognize that the effects of the drought on habitat suitability for these species may manifest over longer timeframes (e.g. large scale pine mortality in southern Sierra). With the continuation of this avian bioregional monitoring program, it will be possible to track the influence of climate related changes on a relatively large number of species (both birds and vegetation) to help inform management responses to these unprecedented changes to the Sierra ecosystem.

Understanding how each of these species responds to changes on the landscape, including management activities, 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 management.

Fox Sparrow

Fox Sparrow occurred at nearly 50% of all point count stations in the study area and occurred on each National Forest unit. However, the species is not evenly distributed across the region. Fox Sparrow occupancy is considerably higher in the central and southern Sierra, but low in 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 2015. While drought may very well be the primary cause of any decline in this species, a number of management actions can alter habitat suitability for Fox Sparrow. These include fire suppression which reults in a slow succession of chaparral and open forest into closed-canopy forest. More immediate are the the short term effects of fire burning chaparral habitat, and the effects of management that focuses on masticating or otherwise reducing shrub density and extent. In the Northern Sierra we found 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).

We continue to find evidence of this species association with dense shrub dominated habitat. Occupancy was higher and extinction lower in areas with higher shrub cover and colonization higher was higher in areas with lower tree cover.

Hairy Woodpecker

Hairy Woodpecker is 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. This species, like most of the woodpeckers in the Sierra Nevada, reaches its greatest density in recently burned forest (Burnett et al. 2012). 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.

In our previous report (Roberts et al. 2015) we showed large variation and uncertainty in occupancy indicating a potential strong, but non-significant, decline over a portion of the time span of our surveys. But new models including one additional year of surveys and more covariate information indicate a steady increase of 1% per year in green forest. As mentioned previously, this species is associated with snags, but in our dynamic occupancy models snags were not selected as a covariate on occupancy, detection probability, or site colonization. However, sites with high snag counts had lower probability of extinction.

Hairy Woodpecker has shown a consistent pattern of increased occupancy or abundance as a result of commercial thinning treatments which do not increase snag density. In the Lassen National Forest we found the species at far greater density in burned forest than mechanically thinned sites (Burnett et al. 2012).

Mountain Quail
Mountain Quail occupancy was remarkably consistent across the six years of our surveys. 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 this species most closely tied to shrub dominated habitats. But, unlike Fox Sparrow they readily occur in the understory of mature open-canopy forest with a shrub component (Roberts et al. 2013). In our dynamic occupancy models both a positive association with shrub cover and a negative association with basal area indicate conditions consistent with early seral forest.

Yellow Warbler
Yellow Warbler occur at over 30% of our riparian point count stations and their occupancy appears to be increasing. The fact that they do not occur at 70% of our sampling stations suggests that their distribution may be limited by both elevation and habitat degradation.

The associations from the dynamic occupancy models are consistent with our understanding of this species habitat requirement and our findings from previous work. Occupancy was higher at sites with higher shrub height, and lower tree basal area. Colonization was higher at sites with high shrub cover and low basal area, and extinction probability increased at sites with less grass cover and less bare ground.

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 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 proxy 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 (e.g. pond and plug) that restores floodplain function and increases the cover of willow 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 seven year data set represents a significant investment on the part of the USFS and Point Blue, and its value will only increase over time as a baseline to assess the combined effects of climate change and management actions well 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 a recent 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). 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). Over 60 locations burned in the 2013 Rim Fire on Stanislaus National Forest and along with adjacent unburned locations, we are currently using these for analyses looking at the avian community in post-fire habitat. We are completing a manuscript

now evaluating the long-term effects of fire and silvicultural treatments on the avian community with this dataset. We plan to continue using the Sierra Nevada bioregional monitoring dataset in future reports, publications, and presentations to help inform forest management and conservation of biodiversity across this ecologically important mountain range.

APPENDIX A: Black-backed Woodpeckers in Green Forest

The Black-backed Woodpecker (*Picoides arcticus*) is an uncommon to rare habitat specialist that reaches its greatest density in moderate and high severity burned forest (Saracco et al. 2011, Hutto 2008). This species also inhabits green forest throughout its range but there are relatively few studies of their life history outside of burned areas. Recently 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 reports (Roberts et al. 2015) 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 2011-2015 on the 474 upland transects located on 10 national forest units (Table A1). We defined green forest as areas that had not burned at moderate or high severity from 1993-2014 and were more than 2 km from areas burned at moderate or high severity from 2004-2014 ($n = 377$ transects). Eleven transects burned between the 2014 and 2015 field seasons and were removed from the dataset (resulting in $n = 366$ total transects).

At each of the five point count stations within a transect we conducted a standardized unlimited distance 5 min 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 min playback survey for Hairy Woodpecker (*Picoides villosus*) and Mountain Quail (*Oreortyx pictus*), and a 6 min 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 min, 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.

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 assembled detection histories for each transect by combining all detections from the five passive point counts during a single transect visit, and considered this as a separate survey event from the playback surveys at the center point. The total time of surveys was different among the survey types, with five, 5 min passive point counts (25 min of passive survey time total per transect) compared to one 5 min Hairy Woodpecker/Mountain Quail playback survey plus one 6 min Black-backed Woodpecker playback survey (11 min of playback survey time per transect). We visited each transect up to twice per year, for a maximum of $K = 4$ survey events per year per

transect. For all models we included survey type (passive or playback) as a covariate of detection probability.

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). Therefore, for each of the $n = 366$ green forest transects there were a maximum of $K = 20$ survey occasions (up to two visits on two separate dates each year, and two survey types per visit). We included the occupancy covariates identified from a separate analysis that we found to most strongly influence Black-backed Woodpecker occupancy (Fogg et al. 2014). These include latitude, elevation, slope, canopy height, and snag density, 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, colonization, and extinction were functions consisting of an intercept term, the occupancy covariates, the year of survey, and detection also included a covariate for survey type, passive [0] or playback [1]. The addition of the full set of covariates with detection probability, colonization, and extinction is different from previous analyses with these data. Model selection proceeded using the standard AIC 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 were detected at green forest transects on all forests except for the Sequoia National Forest and the Lake Tahoe Basin Management Unit (Table A1). They are present on burned forest transects in Lake Tahoe Basin Management Unit, but no detections on burned or unburned transects have occurred on Sequoia National Forest during our surveys (data not shown) but have been detected in the Sequoia in burned forest (Siegel et al. 2015a).

The lowest AIC model following the variable removal process included six occupancy and only two detection covariates (Table A2). Survey type and year of survey were not selected as covariates on detection, colonization, or extinction. Our previous report (Roberts et al. 2015) included year as a covariate on extinction and detection probability a priori and we did not assess their inclusion in the variable selection process. Occupancy is positively associated with higher elevation, northern latitudes, high canopy, northerly aspects, low slopes, and high snag density patches. Detection probability covariates included only latitude and snags, both positively associated with detection. Colonization was higher at high elevations and low slopes, while extinction had no significant variable associations. Notably, year of survey was not included in any of the model components, and thus occupancy estimates vary little across years. The extinction parameter estimate was again high, while colonization was 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 the addition of a fifth year of data has aided in the evaluation of covariates on the extinction and colonization parameters where previously (Fogg et al. 2014 and Roberts et al. 2015) model fit was poor making it difficult to conduct variable selection.

In comparison to our previous report (Roberts et al. 2015), occupancy is much less variable across the years of our surveys (Figure A1). Estimated occupancy ranged from

0.15 in 2011 to 0.19 in 2015 with confidence intervals largely overlapping for all five years. Yearly values for occupancy estimates (and 95% CIs) are as follows: 2011 – 0.15 (0.03-0.26), 2012 – 0.17 (0.10-0.23), 2013 – 0.17 (0.12-0.23), 2014 – 0.18 (0.12-0.23), 2015 – 0.19 (0.13-0.25). Although the occupancy estimates are largely similar to our previous analyses, the pattern among years implies a different interpretation of the trend over time which appears to be stable rather than strongly decreasing as we reported following the 2014 field season (Roberts 2015).

To explain why these occupancy values are lower in comparison to the Fogg et al. (2014) manuscript, transects that were removed prior to the occupancy analysis shown in the 2014 report but were included in the manuscript were primarily lower elevation locations that had a low average estimated occupancy of 0.11. The 30 transects we added prior to the 2014 report analysis were moderate elevation sites with higher average estimated occupancy of 0.19. Of the 11 transects removed due to fires in between the 2014 and 2015 seasons, three had previous detections of Black-backed Woodpeckers. All three of the transects with prior detections were in the Lassen National Forest.

Probability of detection did not include year of survey or survey type as a significant covariate, and therefore it was a single constant value of 0.23 (95% CI = 0.19 – 0.28). Previous reports showed that probability of detection varied strongly; leading, in part, to the correspondingly variable occupancy estimates.

The model estimated low colonization of 0.07 (95% CI = 0.05 – 0.09), and relatively high extinction of 0.44 (95% CI = 0.31 – 0.58), a similar pattern to that found in burned forest in our study area (Siegel et al. 2015a). Error bars for extinction rate estimates are large, though much smaller than in previous reports, and again as we learn more about these methods we feel as though this is 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).

Thus we conclude that differences in occupancy estimates among our various publications regarding Black-backed Woodpecker occupancy are reflections of varying sample datasets and not because of any significant population abundance or distribution trends. Though we cannot rule out the presence of significant trends, this most recent analysis suggests that the distribution of the Black-backed Woodpecker population on actively managed unburned national forest lands in the Sierra bioregion is stable.

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, Chips) during the timeframe of this monitoring project occupancy on our unburned forest sites appear highly stable. 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. 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 per acre).

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
Modoc	6 (44)	7 (44)	3 (46)	6 (46)	8 (46)
Lassen	8 (56)	10 (54)	10 (56)	7 (56)	9 (56)
Plumas	0 (39)	2 (39)	1 (39)	2 (39)	0 (39)
Tahoe	2 (38)	3 (42)	4 (42)	2 (42)	4 (42)
LTBMU	0 (2)	0 (2)	0 (2)	0 (2)	0 (2)
Eldorado	1 (38)	2 (38)	4 (41)	4 (42)	3 (42)
Stanislaus	0 (20)	4 (24)	4 (26)	0 (26)	4 (26)
Inyo	5 (18)	2 (18)	6 (20)	7 (20)	4 (20)
Sierra	5 (56)	4 (56)	6 (60)	5 (60)	6 (60)
Sequoia	0 (29)	0 (29)	0 (33)	0 (33)	0 (33)
Total	27 (340)	34 (346)	38 (365)	33 (366)	38 (366)

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

Black-backed Woodpecker	β	SE	P(>|z|)
Occupancy (ψ)			
Intercept	-3.92	0.81	0.000
Canopy height	1.03	0.40	0.010
Latitude	0.84	0.47	0.075
Elevation	2.34	0.72	0.001
Solar radiation index	-1.08	0.49	0.027
Slope	-1.70	0.57	0.003
Snags	0.65	0.29	0.024
Colonization (γ)			
Intercept	-2.63	0.19	0.000
Elevation	1.34	0.22	0.000
Slope	-0.82	0.18	0.000
Extinction (ε)			
Intercept	-0.23	0.29	0.429
Detection Probability (ρ)			
Intercept	-1.21	0.13	0.000
Latitude	0.17	0.11	0.124
Snags	0.17	0.09	0.056

Figure A1: Annual occupancy estimates (left panel), with colonization and extinction probabilities (right panel), for Black-backed Woodpeckers in green forest. Vertical lines bounding each point indicate 95% confidence intervals. Detection probability, colonization, and extinction were held constant across years in this multi-season occupancy model.
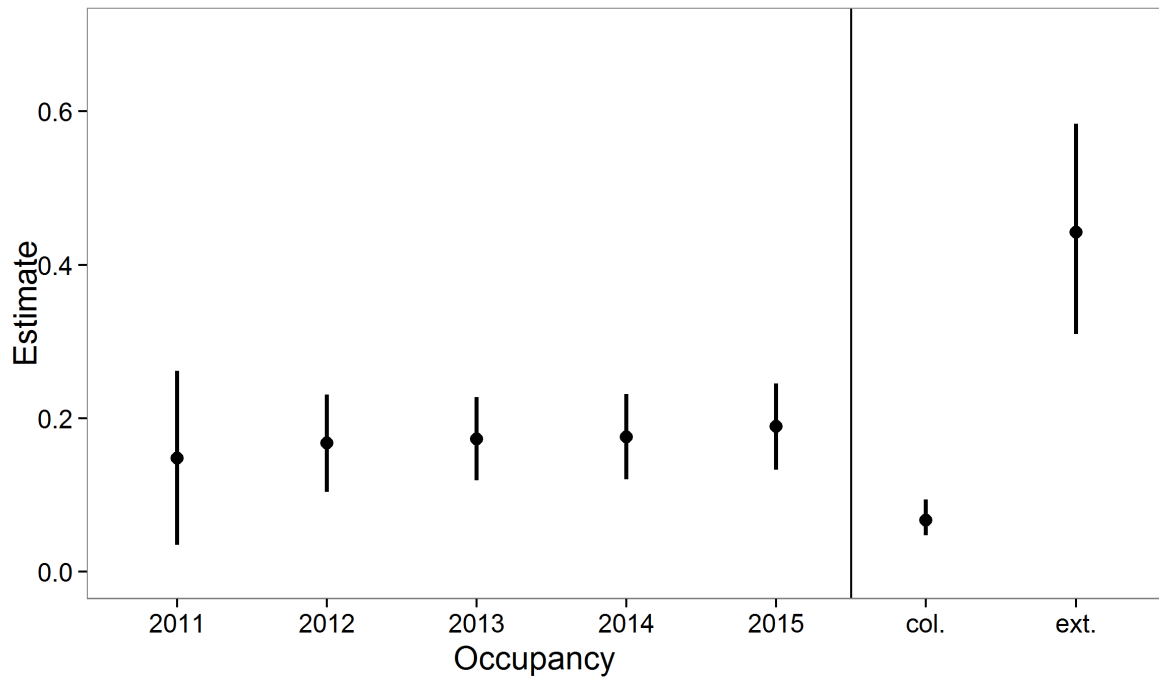
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