prbo

Avian Community Response to Mechanical Fuel Treatment in the Sierra Nevada, USA.

Ryan D. Burnett, Nathaniel E. Seavy, Leo Salas, and Diana L. Humple

PRBO Conservation Science

3820 Cypress Drive #11

Petaluma, CA 94954

PRBO Contribution # 1912

Abstract

We investigated the short term response (2 – 6 years post treatment) of the avian community to three fuels reduction treatments in the Northern Sierra Nevada, USA. We evaluated the effects of shaded fuel breaks, group selections, and pre-commercial understory thinning on the abundance of a suite of focal bird species that serve as surrogate species for three habitat guilds (mature closed canopy forest, shrubs, and edge/open forest). We also measured five vegetation characteristics that we predicted may impact bird habitat in these forest. We found that treatments significantly reduced snag density, overstory and understory tree cover, shrub cover, and herbaceous cover. The effect of treatments on bird abundance was less obvious. The overall abundance of members of each habitat guild was not significantly different between pre and post-treatment conditions for any of the treatments. Species richness modestly increased following shaded fuel break and group selection treatments and significantly decreased after pre-commercial thinning. Species within the habitat guilds often did not respond congruently to treatments and overall effects for most species were modest. However, as predicted mature forest canopy associated species showed the strongest negative responses while edge and open forest species showed the most positive responses to treatment. Our results suggest that these fuel reduction treatments have relatively modest impacts on bird community composition and abundance. The potential of these treatments to benefit the full spectrum of disturbance-dependent species, especially those associated with moderate and high severity fire, appears limited.

Keywords: shaded fuel break, group selection, pre-commercial thin, mastication, songbird, focal species, forest heterogeneity, fire surrogate

1. Introduction

In conifer forests of the Sierra Nevada, fire suppression and silvicultural practices have increased forest stand densities and fuel loads and reduced local and landscape heterogeneity (Taylor, 2004; Beaty and Taylor, 2008; Collins et al., 2011). Because the current structure of many Sierra Nevada forests is believed to increase their vulnerability to stand-replacing fire, managers are increasingly concerned with reducing vulnerability through the mechanical removal of fuels (North et al., 2007; Collins et al., 2010; Safford et al., 2012). These fuels treatments generally involve reducing understory vegetation and thinning medium-sized trees in order to slow the rate at which fires spread, reduce the intensity with which they burn, and increase human safety (Stephens, 1998; Collins et al., 2007).

Because fuels reduction treatments have become one of the primary forest management tools in western North American forests, the degree to which these treatments impact ecological conditions and wildlife habitat is important (Schoennagel and Nelson, 2011; Safford et al., 2012; Stephens et al., 2012). Understanding the ecological consequences of fuels reduction treatments is complicated because they can vary dramatically depending on the intensity and spatial extent of the treatments and because they need to be understood in the context of the natural disturbance regime of the ecosystem in which they are implemented (Fontaine and Kennedy, 2012; Stephens et al. 2012). Monitoring that evaluates effects of these treatments on wildlife habitat and other ecosystem functions is an important element of ensuring treatments can be implemented without risking other ecological values (Lee and Irwin, 2005; Hutto and Belote, 2013).

One approach to ecological monitoring is to focus on groups of organisms that can provide cost-effective information about ecological conditions of interest (Vos et al. 2000; Gram et al. 2001). In this context, birds are an effective tool for monitoring because: (1) many species are easily and inexpensively detected using standardized sampling protocols; (2) these species are sensitive to a wide variety of habitat conditions and their abundance can be used as proxy of habitat quality (Bock and Jones 2004); and (3) accounting for and maintaining many species with different ecological requirements can be used to implement landscape conservation strategies (Hutto 1998). For these reasons, the abundance of bird species previously identified as ecological indicators (i.e., focal species or surrogate species) in habitat conservation plans and other management documents can provide a metric for evaluating the ecological effects of fuels treatments. For example, information about the habitat requirements of Partners in Flight focal species has provided a straightforward way to interpret the ecological effects of fuels treatments in the context of regional conservation priorities (Alexander et al., 2007; Seavy et al., 2008).

Because there are a suite of different fuels reduction prescriptions available to forest managers, information about the relative effects of these treatments on wildlife habitat is relevant to decision-making. Two common fuels treatments prescriptions used in the Sierra Nevada are shaded fuel breaks (also called defensible fuel profile zones) and pre-commercial understory thinning. An additional treatment, group selections, is not employed primarily as a fuels reduction treatment, but was frequently used in our study area in concert with the two other fuel treatments. Because group selections are a common and sometimes controversial silvicultural prescription, we included them in this analysis and refer to all three treatments as fuels reductions.

To evaluate the relative effects of these treatments on wildlife habitat, we selected a suite of passerine birds to use as indicators of ecological change. We propose that the degree to which these treatments altered the forest structure in ecologically meaningful ways could be ascertained from the changes in abundance of these indicator species after treatment (Hutto and Belote, 2013). Because these treatments were generally designed to reduce tree and shrub cover, and increase the amount of forest edge, we predicted that if the treatments were ecologically meaningful, then birds associated with understory shrubs and canopy trees would decline in the initial years post-treatment, whereas those associated with more open canopy and edge would increase. Thus the objectives of this project were to (1) describe the effects of these three fuels treatments on bird abundance and vegetation structure, (2) use these changes to make inferences about the degree to which fuels treatments produce changes in forest structure that are ecologically meaningful for Sierra Nevada wildlife, and (3) provide forest managers with information about the ecological consequences of fuels treatments primarily designed to modify fire behavior.

2. Methods

2.1. Study sites

We studied mechanical fuels reduction treatments in the Lassen and Plumas National Forests within the boundaries of the Herger-Feinstein Quincy Library Group Forest Recovery Act Pilot Project (HFQLG 1999). The study sites encompassed portions of Butte, Lassen, and

Plumas Counties at the intersection of the Sierra Nevada and Cascade mountains of northeastern California, USA (Fig. 1). Survey sites ranged in elevation from 956 to 1896 m within mixed conifer, true fir (*Abies* spp.), and yellow pine (*Pinus ponderosa* and *P. jeffreyi)* cover types.

2.2. Fuel treatment definitions

All of the mechanical treatments we studied were limited to areas where such treatments were allowed under local resource management plans (HFQLG 1999, SNFPA 2004). Thus, treatments generally avoided late-seral habitat, especially where species such as pine marten (*Martes americana)* and spotted owl (*Strix occidentalis*) raise young; streamside zones; steep slopes; and roadless areas. Below, we briefly describe each of the prescriptions, but recognize that implementation sometimes varied depending on local site conditions or personnel.

Shaded fuel breaks were usually adjacent to a forest road, linear, and ranged from 250 – 800 m wide and 1000 – 7000 m long. They were placed in strategic locations intended to reduce predicted fire spread and behavior. Both overstory and understory trees were thinned with a target of reducing canopy cover to 40% and significantly reducing ladder fuels (understory trees and shrubs). The majority of shaded fuel breaks we sampled did not have surface fuels treatments during our study, though at a small number of sites slash was hand-piled and burned. These shaded fuel breaks were all designated as defensible fuel profile zones after implementation.

Group selections were 0.2 – 0.8 ha (0.5 – 2 acres) areas where all or nearly all trees less than 76.2 cm (30 inches) diameter at breast height (DBH) were removed in order to improve conditions for shade intolerant conifer regeneration and increase forest heterogeneity. Some of these sites were replanted with conifers after treatment, though with less than 5 years of post-treatment group selection data included in this analysis, conifer seedlings planted in the years following treatment contributed relatively little to changes in forest structure during the study.

Pre-commercial thins included all treatment types that targeted non-merchantable trees and shrubs. These thinnings were carried out in early successional forest habitats usually by mechanical masticators and to a lesser extent by hand (e.g., chainsaws).

2.3. Sampling design

To evaluate the response of the avian community to the three fuels reduction treatments, we compared bird abundance following treatment to pre-treatment and untreated reference sites. To identify sampling locations that would be treated, we worked with USFS personnel in the initial years of the study to identify 10 projects where fuels treatments were planned. Within these projects, we placed avian point count stations in units that were prioritized for treatment and in units not slated for treatment in the same sub-watersheds.

Following 10 years of point count surveys at these sites, we used the USFS Forest Service Activity Tracking System geo-database and USFS personnel to determine the treatment status of each of our point count stations in each of these projects for each year. We then identified all point count stations for which we had at least one year of pre-treatment bird data and one year of post-treatment data from at least two breeding seasons following treatment. We excluded data from the first breeding season following treatment to reduce the influence of lagged responses to vegetation change (Seavy and Alexander, 2011).

For shaded fuel breaks and pre-commercial thins, we considered a station to be treated if the point count station center was within the footprint of the treatment, no matter how far from the edge of the treatment it was located. Because individual group selections had relatively small footprints (13 – 50 m radius circles), but a potentially large effect (removal of all trees) we considered point count stations that were not within the treatment footprint but fell within 25m of the edge of a group selection as treated. Any sites that received more than one treatment during the study period or were affected by wildfire were excluded from analysis once the second treatment or fire occurred.

Our study was conducted between 2002 and 2011 (10 years). By the end of the study, we had 122 point count stations in shaded fuel breaks (in 18 treatment units and 7 different USFS projects), 49 point count stations in pre-commercial thins (7 treatment units and 4 USFS projects), and 17 point count stations in group selections (8 treatment units and 2 USFS projects; Fig. 1). Treatment of these units was completed between 2002 and 2009, and monitoring data included in this analysis spanned up to eight years for shaded fuel breaks (2004 – 2011) and up to 10 years (2002 – 2011) for group selections and pre-commercial thins. Treatments were not all initiated in the same year, thus not all point count stations were monitored during the same years, such that for any point count station the number of pre-treatment years of data varied from one to five, and the number of post-treatment years varied from two to six.

In addition to using pre- and post-treatment surveys to evaluate changes associated with fuels treatments, we also wanted to compare these points to the natural variability in bird abundance at reference sites that remained untreated during the entire study period. Reference sites were selected from a pool of point count stations that were either identified to be treated but never treated, or were identified as reference sites in stands within the same sub-watersheds as the treated stands. The reference site pool consisted of over 500 points all of which occurred within at least 5km of at least one treated stand. From this pool, we eliminated any point count stations that were <100 m from any fuels treatment implemented between 1999 (the first year for which reliable treatment data was available) and 2011. We also removed points that were <250 m from wildfires that had burned since 1992 (10 years prior to our study). We then used the abundance of 15 avian study species (see below) in a cluster analysis of reference and pre-treatment point count locations. Using the dendrogram generated from the cluster analysis, we identified reference points that had bird communities most similar to those at the pre-treatment points, and then selected a number of reference points that was equal to the number of treatment points for each of the three treatment types.

2.4. Measuring vegetation structure

We collected vegetation data across a 50-m radius plot centered on the point count station. On these plots we made ocular estimates of the percent cover of overstory trees (>5 m), understory trees (<5 m), shrubs, and herbaceous vegetation, and counted all snags in two DBH size categories (10-30 cm and > 30 cm). We collected vegetation data at all 122 shaded fuel break point count stations both pre- and post-treatment; 38 of 49 pre-commercial thin stations pre-treatment and 29 post-treatment; 17 of 17 group selection stations pre-treatment and 9 post-treatment; and all reference stations for each treatment. Vegetation data were collected at treated sites 1 to 4 years prior to treatment and 1 to 4 years following treatment, and at reference sites once at the beginning of the study from 2003-2005.

2.5. Measuring bird abundance

The avian community was sampled using five-minute point count surveys (Reynolds et al., 1980; Ralph et al., 1995). In this method points are clustered in transects, but data are only collected at the individual point. All birds detected at each point during the five-minute survey were recorded according to their initial distance from the observer. All observers underwent an intensive, three-week training period focused on bird identification and distance estimation prior to conducting surveys, and laser rangefinders were used to assist in distance estimation at every survey point. Surveys began around local sunrise, were completed within four hours, and did not occur in inclement weather (rain, snow, or wind > 20kph). All sites were surveyed two times in each year and observers were rotated between visits within years. All point count stations were at least 230 m apart.

2.6. Ecological inference

We evaluated changes in vegetation structure that occurred as a result of the treatment to verify that these changes were generally consistent with the desired effects of the fuels reduction prescriptions, and to compare the variability of treated sites to the variability in forest structure at reference sites.

We identified a suite of 15 bird species to use to evaluate the ecological effects of fuels treatments. The species were chosen because they (1) were detected in sufficient numbers to allow for meaningful statistical analysis (Nur et al. 1999), (2) were appropriately sampled using the standardized point count survey method (Ralph et al. 1993), and (3) represented a range of ecological conditions found throughout our study area (Chase and Guepel 2005). To select these species, we began with the California Partners in Flight Coniferous Forest Bird Conservation Plan (CalPIF 2002), and then supplemented this list with seven additional species that complemented the CalPIF focal species. The list included permanent residents, short-distance migrants, and Neotropical migrants; and ranged in habitat associations from mature dense canopy forest species to early successional chaparral species.

Based on our local knowledge and published information about the habitat associations of these 15 species, we grouped them into three guilds representing shared associations with forest structure and composition. The forest canopy guild species were associated with relatively mature forest stands with considerable canopy closure: Golden-crowned Kinglet (*Regulus* *satrapa*), Hermit Warbler (*Setophaga occidentalis*), Red-breasted Nuthatch (*Sitta canadensis*), Hammond's Flycatcher (*Empidonax hammondii*), and Brown Creeper (*Certhia americana*). The understory guild species were associated with low-growing foliage (broadleaf shrubs and herbaceous species) and often more open stand conditions: Dark-eyed Junco (*Junco hyemalis*), Dusky Flycatcher (*Empidonax oberholseri*), Fox Sparrow (*Passerella iliaca*), Nashville Warbler (*Oreothlypis ruficapilla*), and MacGillivray's Warbler (*Geothlypis tolmiei*). Finally, the forest edge guild species were those that occur primarily along forest edges or in areas with breaks in the canopy: Olive-sided Flycatcher (*Contopus cooperi*), Western Tanager (*Piranga ludoviciana*), Chipping Sparrow (*Spizella passerina*), Mountain Chickadee (*Poecile gambeli*), and Yellow-rumped Warbler (*Setophaga coronata*).

We hypothesized that if the treatments had ecologically meaningful effects for birds, then species in the canopy guild would respond negatively to all three fuels reduction treatments, species in the edge guild would respond positively to all three treatments, and species in the understory guild would respond negatively to shaded fuel breaks and especially pre-commercial thinning, and have a neutral or positive response to the group selection treatment.

Finally, we used the number of species detected at each point as a measure of species richness to evaluate the effects of fuels treatments on the avian community. Because managers are often encouraged to manage for diversity, we wanted to be able to ask if on average there were more or fewer bird species detected at a point after fuels treatments were implemented. We hypothesized that if treatments provided greater structural heterogeneity in vegetation, that the number of bird species per point would increase, whereas if they created greater homogeneity in vegetation structure, the number of species per point would decrease (Verner and Larson 1989).

2.7. Data analysis

To analyze changes in vegetation structure, we evaluated the difference between pre and post-treatment vegetation using a two-tailed student t-test with unequal variance for each of the five vegetation metrics. Although multiple tests inflate the table-wide error rate (Rice 1989), we chose not to use Bonferroni corrections in our interpretation of p-values, following instead the recommendations of Moran (2003): we report exact p-values, used an uncorrected α of 0.05, and interpret the change in vegetation in the context of ecological significance to the bird species of interest.

For our analysis of changes in bird abundance, we were initially concerned that detectability may have varied among treatments. To evaluate detectability, we began by fitting an imperfect-detection mixture model that considered a single Poisson mean of abundance and a single probability of detection per treatment level (Zuur et al., 2009) using package “pscl” (Zeileis et al., 2008) in program R (R Core Development Team, 2011). With this model, we assumed repeated visits to a point generated independent estimates of density for the treatment level. This assumption had little bearing on our analyses, since its main consequence would be that of underestimating the variance around the parameters in the model (Oksanen, 2001; Millar and Anderson, 2004), in particular variance around the probability of detection. These models showed substantial congruence in estimates of probability of detection between treatment levels for all species in our study (none showed statistically significant differences, see Appendix A). Thus, even with underestimated variances, the model failed to detect any differences in probability of detection among treatment conditions. Because we found little evidence for substantial variation in detection probabilities among treatments, for our final analysis we fitted a simpler random-effects generalized linear model to the data using the R package “lme4” (Bates et al., 2011), instead of opting for the more complex hierarchical imperfect detection models (Royle, 2004). These random-effects generalized linear models were more appropriate for our study design than the initial imperfect-detection mixture models because they allowed us to use random-effects to accommodate the spatially-nested structure of our data.

We initially were interested in evaluating the effect of time since treatment on the avian community but preliminary analysis revealed few patterns and analyzing each year separately significantly reduced our power to detect effects. Thus, we considered the overall treatment effects by pooling data across all post-treatment years and controlling for annual variation.

In our final models we considered counts (the response parameter) to be the maximum number of detections within 50 m of the point across the two surveys at a point in a year. We recognized that this metric reflects both true densities at a point and the probability of detecting all individuals given that a species is present (Royle and Nichols, 2003), but given the lack of evidence that detectability varied among treatments, it provides a useful index of true density of these species (Johnson 2008).

Because the maximum counts were discrete, we used a Poisson link function for generalized linear model with random and fixed effects using “lme4” package (Bates et al.,

2011) in R. Random-effect models are well suited to capture nested designs such as ours, where survey points are clustered in space (ie transects; Pinheiro and Bates, 2000). Random effects on the intercept parameter included year and the factor transect:point, or year, transect and point (i.e., two competing models). The random effect of year on the intercept captures the variation across years in density, the random effect due to transect captures regional differences in density, and the point effect captures differences among points. The factor transect:point captures the differences among points but within the nested relationship of points within transects (Pinheiro and Bates, 2000). We included a single fixed effect that categorized the condition of point as reference, pre-treatment, or post-treatment. We chose the model that best fit the data by value of AIC coefficient.

In addition to analyzing the abundance of the 15 species individually, we also evaluated the overall effect of these treatments on each guild by considering species as a random effect. Finally, to evaluate the effects of treatments on bird richness (i.e., number of species) at a point, we used a random-effect models as described above, but with response parameter being the maximum number of species recorded at a point in a year across both visits.

Although the mixed-effects models produce estimates and standard errors for the effects of treatment levels, we did not consider analysis of differences between these estimates. Nevertheless, some patterns are evident and we illustrate these through plots of mean values and confidence limits contrasting treatment effects.

We evaluated the evidence for an effect of treatment based on three factors: 95% statistical significance between treated and pre-treatment (confidence intervals did not overlap means), the magnitude of change from pre to post-treatment, and by comparing post-treatment and reference site means and confidence intervals.

3. Results

3.1. Vegetation structure

All five vegetation characteristics we measured were lower in post-treatment than pre-treatment or untreated sites. Across all treatments the most consistent change in vegetation structure following treatment was a reduction in snags. In shaded fuel breaks the number of large snags (>30 cm DBH) was reduced by 50% following treatment (P = 0.009; Fig. 2). Similarly, group selections (P = 0.03) saw a significant reduction in large snags. Snags were rarest pre-treatment in pre-commercial thins and were reduced modestly following treatment ($P = 0.61$). After treatments were implemented, large snag density was highest in shaded fuel breaks (4.2/ha) and lowest in group selections (2.2/ha). Small snags showed similar patterns. Shaded fuel breaks had significantly less overstory ($P < 0.001$) and understory tree cover ($P < 0.001$) following treatment, shrub cover showed a modest decrease ($P = 0.05$), and herbaceous cover showed little change (Fig. 2). In group selections, we observed some of the largest changes in vegetation structure, with most being significantly different from pre-treatment conditions. These treatments showed the largest percent change in overstory ($P < 0.001$) and understory tree cover ($P = 0.002$) and herbaceous cover ($P = 0.05$) of any treatment (Fig. 2). There was little evidence of a difference in shrub cover following treatment (Fig. 2). Understory tree ($P = 0.002$), shrub ($P < 0.001$), and herbaceous cover ($P = 0.03$) all decreased significantly following pre-commercial thinning, while overstory tree cover showed only a small decrease (Fig. 2).

To summarize, we observed substantial changes in vegetative structure with the majority of metrics showing over 30% reduction following treatment. Post-fuel reduction conditions across each of these treatments were characterized by low snag densities, low (group selection) to moderate canopy cover, little understory tree cover, moderate shrub cover, and little herbaceous cover. In terms of changes from pre-treatment conditions we found the effects of treatment on vegetation structure strongest in group selections, followed by pre-commercial thins, and weakest in shaded fuel breaks.

3.2. Bird response to treatments

The fifteen species we selected to evaluate the effects of fuels treatments on the avian community accounted for 82% of all bird detections within our 50 m radius sample across treatment types and years, and included the ten most detected species. Among our focal species, Dusky Flycatcher represented the largest proportion with 11% of all detections while Chipping Sparrow was the smallest proportion accounting for 0.45% of all detections.

When we combined species into habitat guilds, in shaded fuel breaks the abundance of each guild showed little change as a result of treatment; in group selections there was weak evidence of a decrease for canopy species and an increase for understory species, and no change for edge species as a result of the treatment; in pre-commercial thins there was weak evidence for a decrease in the abundance of canopy and understory species and no change for edge species as a result of the treatment (Fig. 3). Across all guilds and treatments there was substantial overlap of the 95% confidence intervals (Fig. 3).

In post-treatment shaded fuel breaks the abundance of 14 of the 15 focal species was equal to or greater than prior to treatment, with five species showing a significant increase from pre to post treatment condition (Fig. 4): one mature forest (Brown Creeper), two understory (Fox Sparrow and Dark-eyed Junco), and two edge species (Olive-sided Flycatcher and Western Tanager). Golden-crowned Kinglet was the only species less abundant at treated sites, and this difference was small and not statistically significant. At treated sites, the abundance of species in the edge and understory guilds were equal to or greater than at reference points, while the canopy guild species were equal to or lower than at reference points; the exception was Brown Creeper, which was far more abundant at post-treatment points.

In group selections, the abundance of 8 out of 14 species was equal or greater following treatment (Fig. 5). We were unable to produce a meaningful comparative model for the 15th species, Olive-sided Flycatcher, due to a near absence of records at reference and pre-treatment group selection points, although their results are included (Fig. 5). We interpreted our results as showing five species responding positively and four negatively to group selections. Of these, three positive responses (Dark-eyed Junco, Fox Sparrow, and Chipping Sparrow) and one negative response (Brown Creeper) were statistically significant. One edge and four understory associates were among those with a positive response while all four species with a negative response were mature forest-closed canopy associates. The abundance of mature forest species was generally lower at post-treatment points than reference, understory species were generally more abundant at post-treatment points than reference, and edge species showed no consistent pattern.

For the pre-commercial thins we interpreted the response of three species to be positive and five to be negative, of which all but one (MacGillivray’s Warbler) was statistically significant (Fig. 6). All three of the positive responses were species associated with open forest and edges, while those having a negative response included three mature forest-closed canopy species (Brown Creeper, Hammond’s Flycatcher, and Hermit Warbler) and two understory species (MacGillivray’s Warbler and Nashville Warbler). Despite the apparent decrease in abundance following treatment, the mature forest species were all more abundant at post-treatment points than reference points, as were all of the edge species, while two understory species were more abundant in post-treatment points and three were less abundant than reference points.

In post-treatment sites overall avian species richness was highest in group selections and lowest in shaded fuel breaks. Richness showed a modest non-significant increase in shaded fuel breaks and group selections following treatment and a significant decrease at pre-commercial thins (Fig. 7).

4. Discussion

4.1. Overview

Mechanical fuels reduction treatments, as employed in our study area, appear capable of altering the suitability of habitat for conifer-forest-breeding passerine bird species, in both negative and positive directions depending on the species, species guild, or treatment type. Though there are several limitations to this study, due to its relatively large sample size and geographic scope it fills an important gap in information about the effects of fuel treatments on a group of wildlife species that occupy a broad range of habitat conditions in the Sierra Nevada ecosystem and contributes to the growing body of literature on the effects of fuels treatments on forested ecosystems across North America.

4.2 Ecological effects of fuels treatments

The changes in avian community metrics we evaluated were less extreme than we expected based on changes observed in the forest structure we observed, both in terms of statistical significance and overall effect size.

In our analysis of avian guilds, we generally found little support for an ecological effect of the treatments given the modest and uncertain response of the species combined. This is consistent with a recent meta-analysis of the effects of fuel treatments on birds and supports their conclusion to be cautious in assuming consistent responses to these treatments among species of with similar habitat associations (Fontaine and Kennedy, 2012). The closed canopy-mature forest guild showed the most consistent responses of any guild. They responded negatively to group selection and pre-commercial thins as we predicted, but had a neutral to slightly positive response to shaded fuel break treatments, the latter contrary to our prediction. In fact, one species, the Brown Creeper, showed the strongest positive response to shaded fuel break treatment of any species, nearly doubling in abundance following treatments.

However, when individual species were considered, in general decreases in the abundance of mature-forest closed-canopy associates, and increases in the abundance of species associated with edge and open forest conditions, provide evidence of an ecological effect on the bird community that is consistent with the observed changes in forest structure.

Of the treatments we investigated, group selections, though small in size, appear to have the most substantial effects on the avian community. Consistent with our predictions, the closed canopy guild responded negatively and edge species had a neutral to positive response. But, contrary to our predictions, the understory guild responded positively even though we did not detect a change in shrub cover after treatment. It is possible that the species composition of shrubs changed – away from ground-cover species to the more erect plant species that supply the preferred nesting structure for many of these species.

4.2. Caveats and limitations

This study investigated the short-term effects (2 – 6 years post-treatment) of fuel reduction activities, and thus provides an incomplete picture of treatment effects on breeding landbirds. Post-treatment successional processes may result in considerable change at these sites over longer time periods, though recent evidence suggests that at least shaded fuel breaks in the Northern Sierra exhibit little vegetative change in at least the first 10 - 15 years following treatment (L. Chiono pers. comm.).

The results of this study should also be considered in the context of the conditions that existed in the study area prior to implementation of these treatments as the entire study area had been subjected to a century of timber harvest and fire suppression.

Our analysis was focused primarily on species that are fairly common to abundant. We lacked sufficient data for rarer species to conduct meaningful analysis of the effects of these treatments on their abundance. For the two rarer species we did include (Olive-sided Flycatcher and Chipping Sparrow) error around estimates is large and in one case we were unable to fit a model, thus justifying our exclusion of even less common species. It is possible the species that are most sensitive to silvicultural treatments may be the less common ones in these forests that have been actively managed with mechanical treatments for over a century. However, the avifauna of the Sierra Nevada is still rather intact with no upland species in our study area listed as threatened or endangered. We hypothesize that the upland avifauna of the Sierra Nevada are adapted to regular disturbance. Other studies in western North American forests have shown that few if any landbird species appear to be negatively affected by fragmentation or habitat edges (McGarigal and McCombs, 1995; Scheick et al., 1995; Tewskbury et al., 1998, 2006; George and Dobkin, 2002). The consistent large reduction in snags we observed as a result of these treatments has the potential to affect woodpecker species, though other studies have shown primarily positive fuels treatment effects for most woodpecker species (Fontaine and Kennedy, 2012).

Our group selection sample size was small (17 points) and limited to a relatively small portion of our overall study area. Our power to detect effects was small and caution is advised in extrapolating the effects we did find to other portions of the Sierra Nevada. Further study of the effects of these treatments on the avian community in the Sierra Nevada is needed.

Finally, it is important to consider that this study only investigated the abundance patterns of species and not demographic parameters (productivity or survival). In some systems, the abundance (or density) of birds may (Bock and Jones, 2004) or may not (Van Horne, 1983) be a good estimate of the viability of habitat for a species, in particular when it may take longer than the span of our study to detect changes in abundance driven by changes in productivity, survival, and the immigration of birds from surrounding source areas (Sallabanks et al., 2000; Griesser et al. 2007).

4.3. Management implications

If an objective of these treatments is to minimize negative impacts to wildlife, they appear to have been fairly successful, as the magnitude of difference for most species between pre and post-treatment were relatively modest and all species present prior to treatment were present following treatment. However, a number of disturbance-dependent species were rare in pre-treatment sites and, though some may have increased, they were still quite rare following treatment (e.g. Fox Sparrow & Olive-sided Flycatcher). A frequently-stated objective for mechanical fuels reductions is to mimic the effects of natural disturbance in forested ecosystems (Fontaine and Kennedy 2012, Stephens et al. 2012). As such, an approach that considers the strategic placement of fuel treatments outside of late successional closed-canopy habitats (e.g.

spotted owl home range core areas) and designs fuel reductions to benefit disturbance dependent species in western forest may be a more prudent approach to manage for biological diversity (Hurteau et al. 2007, Fontaine et al. 2009). A number of the open forest and understory species we investigated, as well as a number of other species too rare in our study area to evaluate (e.g. Lazuli Bunting and Mountain Bluebird), reach their greatest abundance in post-fire habitat. These treatments, at least in the short term, appear ineffective in creating habitat that supports equivalent densities of these species as in areas burned in wildfires in the region based on results of an ongoing study in the region (PRBO data), and reinforces the results of a recent meta-analysis of the effects of fire and fire surrogate treatments on birds (Fontaine and Kennedy 2012).

Most of the shrub-nesting species we evaluated were uncommon in shaded fuel break treatments prior to and following treatment. In order to more effectively mimic the mosaic patterns created through natural disturbance and benefit a greater number of species dependent upon those disturbances we suggest - where appropriate – shaded fuel break treatments that consider a greater variation in canopy cover (Chambers et al., 1999; Hagar et al., 2004; Siegel and DeSante, 2003). Mechanical treatments that significantly reduce canopy cover or create canopy gaps can result in increased abundance of middle- and understory-associated landbirds in western forests and overall avian diversity (Hansen et al., 1995; Siegel and DeSante, 2003; Hagar et al., 2004). Additionally, many forest-interior associated birds may benefit from small gaps in mature forest as they utilize the unique resources those gaps provide such as nectar, fruit, seeds and deciduous associated insects (Vitz and Rodewald, 2006; Greenberg et al., 2007; Altman and Hagar, 2007). We found a modest negative effect of group selection treatments on mature forest birds thus the benefits of forest openings to these species may manifest over longer time periods after understory vegetation has been re-established; further study is needed to determine the long-term value or detriment of small forest openings to closed canopy associated species in the Sierra Nevada.

The importance of forest structural diversity for landbirds in western forests is well established (Beedy, 1981; Verner and Larson, 1989; Wilson and Comet 1996). Fuel treatments that remove structure can have negative impacts on a number of avian species while benefiting relatively few (Rodewald and Smith 1998), as we found with pre-commercial thinning. Interestingly, the mature forest species had the strongest negative response to these treatments along with the ground nesting and hardwood associated Nashville Warbler. Variable canopy cover and understory retention in both pre-commercial thins and shaded fuel breaks should allow for greater structural diversity by invigorating or maintaining shade intolerant understory plant assemblages utilized by a large number of avian species in these forests (Hagar et al., 2004).

Since treating surface fuels through prescribed fire following mechanical treatments has been shown to be more effective fuel treatment than mechanical treatments alone (Stephens and Moghaddas, 2005; Stephens et al., 2009), it is unfortunate we were not able to evaluate the effects of fuels treatments and prescribed fire on the avian community in our study. The fact that we had a very small sample of sites that had been burned suggests the limited application of prescribed fire in these forests. Fontaine and Kennedy (2012) recently summarized effects of these combined treatments across fire prone forests of North America but they did not present results showing effects of these combined treatments for any of our study species. They found that responses to mechanical-plus-fire treatments had greater effects than either treatment alone, but they found no interactions where a species response switched (e.g. from positive to negative) following the addition of fire to mechanically treated stands. Studies should be conducted evaluating the combined effect of these treatments as prescribed burning following mechanical treatments becomes more commonplace in the Sierra Nevada (Safford et al. 2012).

5.1 Conclusions

Management decisions should be made in the context of current trends in forest structure and disturbance patterns in order to strike a balance that ensures the full complement of habitat types and structural characteristics to support the range of biological diversity dependent on fire-adapted western forests systems. The goal of land management may not always be to maximize the number of species that benefit from a treatment while minimizing those that do not. Such a species-richness oriented approach may lead to more homogenization of the landscape and declines in species with particular ecological needs. We suggest that a landscape-based ecological approach is prudent, promoting an increase in late successional habitat in some locations while prescribing greater reductions in canopy cover that mimic natural disturbance patterns in areas where biological diversity is relatively low (e.g. closed canopy mid successional stands). Under current management strategies being implemented on National Forest lands in the Sierra Nevada, the loss of late-seral forest, landscape heterogeneity, and fire-dependent habitats appear to be the greatest threat to biodiversity. A balanced approach using a full range of management tools and prescriptions is advisable to ensure biodiversity is sustained.

Acknowledgements

Funding for this project was provided by the Lassen and Plumas National Forests and a grant from the Resources Legacy Fund's Western Conservation Program. Specifically we thank Colin Dillingham of the Plumas National Forest, Coye Burnett of the Lassen National Forest, Peter Stine the USFS National Coordinator for Experimental Forests and Ranges, and Rick Bottoms of the USFS Pacific Southwest Research Station for their support and logistical assistance with this project over many years. We are also grateful to the dozens of field biologists who collected all of the data used in this manuscript. This is PRBO contribution number 1912.

References

Alexander, J.D., Seavy, N.E., Hosten, P., 2007. Using bird conservation plans to evaluate ecological effects of fuels reduction in southwest Oregon oak woodland and chaparral. Forest Ecology and Management 238, 375–383.

Altman, B. Hagar, J., 2007. Rainforest birds: a land managers guide to breeding bird habitat in young conifer forests in the Pacific Northwest. U.S. Geological Survey, Scientific Investigations Report 2006 – 5304.

Beaty, R.M., Taylor, A.H., 2008. Fire history and the structure and dynamics of a mixed conifer forest landscape in the northern Sierra Nevada, Lake Tahoe Basin, California, USA. Forest Ecology and Management 255, 707-719.

Bates, D., Maechler, M., Bolker, B. 2011. lme4: Linear mixed-effects models using S4 classes. R package version 0.999375-42.

Beedy, E.C. 1981. Bird communities and forest structure in the Sierra Nevada of California. The Condor 83, 97-105.

Bock C.E., Jones, Z.F., 2004. Avian habitat evaluation: should counting birds count? Front. Ecol. Environ. 2, 403–410

CALPIF (California Partners in Flight). 2002. Version 1.0. The coniferous forest bird conservation plan: a strategy for protecting and managing coniferous forest habitats and associated birds in California (J. Robinson and J. Alexander, lead authors). Point Reyes Bird Observatory, Stinson Beach, CA. http://www.prbo.org/calpif/plans.html.

Chambers, C.L., McComb, W.C., Tappeiner, J.C., II., 1999. Breeding bird responses to three silvicultural treatments in the Oregon Coast Range. Ecological Applications 9, 171-185.

Chase, M.K., Geupel, G.R., 2005. The use of avian focal species for conservation planning in California. In C.J. Ralph and T.D. Rich (eds.), Bird Conservation Implementation and Integration in the Americas: Proceedings of the Third International Partners in Flight Conference. USDA Forest Service Publication, Gen. Tech. Rep. PSW-GTR-191, Albany, CA.

Collins, B.M., Moghaddas, J.J., Stephens, S.L., 2007. Initial changes in forest structure and understory plant communities following fuel reduction activities in a Sierra Nevada mixed conifer forest. Forest Ecology and Management 239, 102-111.

Collins, B.M., Stephens, S.L., Moghaddas, J.J., Battles, J., 2010. Challenges and approaches in planning fuel treatments across fire-excluded forested landscapes. Journal of Forestry 108, 24-31.

Collins, B.M., Everett, R.G., Stephens, S.L., 2011. Impacts of fire exclusion and recent managed fire on forest structure in old growth Sierra Nevada mixed-conifer forests. Ecosphere 2, 1-14.

Fontaine, J.B., Kennedy, P.L., 2012. Meta-analysis of avian and small-mammal response to fire severity and fire surrogate treatments in US fire-prone forests. Ecological Applications 22, 1547-1561.

Hagar, J., Howlin, S., Ganio, L., 2004. Short-term response of songbirds to experimental thinning of young Douglas-fir forests in the Oregon Cascades. Forest Ecology and Management 199, 333-347.

Hansen, A.J., McComb, W.C., Vega, R., Raphael, M.G., Hunter, M., 1995. Bird habitat relationships in natural and managed forests in the west cascades of Oregon. Ecological Applications 5, 555-569.

HFQLG (Herger-Feinstein Quincy Library Group Forest Recovery Act) 1999. Final Environmental Impact Statement, U.S. Department of Agriculture, Forest Service, Pacific Southwest Region, Vallejo, CA. http://www.fs.fed.us/r5/hfqlg/publications/1999_feis/TOC.htm

Hurteau, S.R., Sisk, T.D., Block, W.M., Dickson, B.G., 2008. Fuel-reduction treatment effects on avian community structure and diversity. Journal of Wildlife Management 72, 1168-1174.

Hutto, R.L., 1998. Using landbirds as an indicator species group. In: Marzluff, J.M., Sallabanks, R. (Eds.), Avian Conservation: Research and Management. Island Press, Covelo, California, pp. 75–92.

Hutto R.L. and R.T. Belote. 2013. Distinguishing four types of monitoring based on the questions they address. Forest Ecology and Management 289:183-189.

George, T.L, Dobkin, D.S., (eds.). 2002. Effects of habitat fragmentation on birds in western landscapes: Contrasts with paradigms from the eastern U.S. Studies in Avian Biology 25, 270 p.

Gram, W.K., Sork, V.L., Marquis, R.J., Renken, R.B., Clawson, R.L., Faaborg, J., Fantz, D.K., Le Corff, J., Lill, J., Porneluzi, P.A., 2001. Evaluating the effects of ecosystem management: a case study in Missouri Ozark forest. Ecol. Appl. 11, 1667–1679.

Greenberg, C.H., Tomcho, A.L., Lanham, J.D., Waldrop, T.A., Tomcho, J., Phillips, R.J., Simon, D., 2007. Short-term effects of fire and other fuel reduction treatments on breeding birds in a Southern Appalachian upland hardwood forest. Journal of Wildlife Management 71, 1906-1916.

Griesser, M., Nystrand, M., Eggers, S., Ekman, J., 2007. Impact of forestry practices on fitness correlates and population productivity in an open-nesting bird species. Conservation Biology, 21, 767-774.

Johnson, D.H. 2008. In defense of indices: the case of bird surveys. Journal of Wildlife Management 72, 857-868.

Lee, D.C., Irwin, L.L., 2005. Assessing risks to spotted owls from forest thinning in fire-adapted forests of the western United States. Forest Ecology and Management 211, 191-209.

Millar, R.B., Anderson, M.J., Remedies for pseudoreplication. 2004. Fisheries Research 70, 397-407.

McGarigal, K., McComb, W.C., 1995. Relationships between landscape structure and breeding birds in the Oregon Coast Range. Ecological Monographs 65, 235-260.

Nur, N., Jones, S.L., Geupel, G.R., 1999. Statistical guide to data analysis of avian monitoring programs. U.S. Fish & Wildlife Service, Biological Technical Publication R6001-1999.

Oksanen, L. 2001. Logic of experiments in ecology: is pseudoreplication a pseudoissue? Oikos 94, 27–38.

Pinheiro, J.C., Bates, D.M., 2000. Mixed-effects models in S and S-PLUS. Springer Verlag.

R Development Core Team. 2009. R: a language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria.

Ralph, C.J., Geupel, G.R., Pyle, P., Martin, T.E., and DeSante, D.F., 1993. Field Methods for Monitoring Landbirds. U.S. Department of Agriculture, Forest Service, General Technical Report PSW-144.

Ralph, C.J., Droege, S., Sauer, J.R., 1995. Managing and monitoring birds using point counts: standards and applications. In C. J. Ralph, J. R. Sauer and S. Droege (eds.), Monitoring Bird Populations by Point Counts. USDA Forest Service Publication, Gen. Tech. Rep. PSW-GTR-149, Albany, CA .

Reynolds, R.T., Scott, J.M., Nussbaum, R.A., 1980. A variable circular plot method for estimating bird numbers. Condor 82, 309:313.

Rodewald, P.G., Smith, K.G., 1998. Short-term effects of understory and overstory management on breeding birds in Arkansas oak-hickory forests. The Journal of Wildlife Management 62, 1411-1417.

Royle, J.A. 2004. N-mixture models for estimating population size from spatially replicated counts. Biometrics 60: 108-115.

Royle, J.A., Nichols, J.D., 2003. Estimating abundance from repeated presence-absence data or point counts. Ecology 84, 777-790.

Safford, H.D., Stevens, J.T., Merriam, K., Meyer, M.D., Latimer, A.M., 2012. Fuel treatment effectiveness in California yellow pine and mixed conifer forests. Forest Ecology and Management 274, 17-28.

Sallabanks, R., Arnett, E.B. Marzluff, J.M., 2000. An evaluation of research on the effects of timber harvest on bird populations. Wildlife Society Bulletin 28, 1144-1155.

Schieck, J.K., Lertzman, B., Nyberg, B., Page, R., 1995. Effects of patch size on birds in old-growth montane forests. Conservation Biology 9, 1072-1084.

Schoennagel, T., Nelson, C.R. 2011. Restoration relevance of recent National Fire Plan treatments in forests of the western United States. Front. Ecol. Environ 9, 271-277.

Seavy, N. E., Alexander, J. D., 2011. Interactive effects of vegetation structure and composition describe bird habitat associations in mixed broadleaf–conifer forest. Journal of Wildlife Management, 75, 344-352.

Seavy, N. E., Alexander, J.D., Hosten, P.E., 2008. Bird community composition after mechanical mastication fuel treatments in southwest Oregon oak woodland and chaparral. Forest Ecology and Management 256, 774-778.

Siegel, R.B., DeSante, D.F., 2003. Bird communities in thinned versus unthinned sierran mixed conifer stands. Wilson Bulletin 115, 155-165.

Stephens, S.L, Moghaddas, J.J., 2005. Experimental fuel treatment impacts on forest structure, potential fire behavior, and predicted tree mortality in a California mixed conifer forest. Forest Ecology and Management 215, 21-36.

Stephens, S.L., Moghaddas, J.J., Edminster, C., Fiedler, C.E., Haase, S., Harrington, M., Keeley, J.E., Knapp, E.E., McIver, J.D., Metlen, K., Skinner, C.N., Youngblood, A., 2009. Fire treatment effects on vegetation structure, fuels, and potential fire severity in western U.S. forests. Ecological Applications 19, 305-320.

Stephens, S.L., McIver, J.D., Boerner, R.E.J., Fettig, C.J., Fontaine, J.B., Hartsough, B.R., Kennedy, P.L., Schwilk, D.W., 2012. The effects of forest fuel-reduction treatments in the United States. Bioscience 62, 549-560.

Stine, P., Landram, M., Keane, J., Lee, D., Laudenslayer, B., Weatherspoon, P., and Baldwin, J. 2002. Fire and fuels management, landscape dynamics, and fish and wildlife resources: An integrated research plan for the Herger-Feinstein Quincy Library Group Pilot Study Area. http://www.fs.fed.us/psw/programs/snrc/forest_health/plas_studyplan.pdf

Tewksbury, J.J., Hejl, S.J., Martin, T.E., 1998. Fragmentation in a western landscape: forest fragmentation does not reduce nesting success. Ecology 79, 2890-2903.

Tiedemann, A.R., Klemmedson, J.O., Bull, E.L., 2000. Solution of forest health problems with prescribed fire: are forest productivity and wildlife at risk? Forest Ecology and Management 127, 1–18.

Vanderwel, M.C., Malcom, J.R., Mills, S.C., 2007. A meta-analysis of bird responses to uniform partial harvesting across North America. Conservation Biology 21, 1230-1240.

Van Horne, B. 1983. Density as a misleading estimate of habitat quality. The Journal of Wildlife Management 47, 893-901.

Verner, J., Larson, T.A., 1989. Richness of breeding bird species in mixed-conifer forests of the Sierra Nevada, California. Auk 106, 447-463.

Vitz, A.C. Rodewald, A.D., 2006. Can regenerating clearcuts benefit mature-forest songbirds? An examination of post-breeding ecology. Biological Conservation 127, 477-486.

Vos, P., Meelis, E., Ter Keurs, W.J.A., 2000. A framework for the design of ecological monitoring programs as a tool for environmental and nature management. Environ. Monit. Assess. 61, 317–344.

Wayman R.B. North, M., 2007. Initial response of a mixed-conifer understory plant community to burning and thinning restoration treatments. Forest Ecology and Management 239, 32-44.

Wilson, M..F., Comet, T.A., 1996. Bird communities of northern forests: ecological correlates of diversity and abundance in the understory. Condor 98, 350-362.

Zeileis, A., Kleiber, C., and Jackman, S. 2008. Regression Models for Count Data in R. Journal of Statistical Software 27, 1-25.

Zuur, A.F., Ieno, E.N., Walker, N., Saveliev, A.A., Smith, G.M., 2009. Mixed effects models and extensions in ecology with R, first ed. New York, New York.

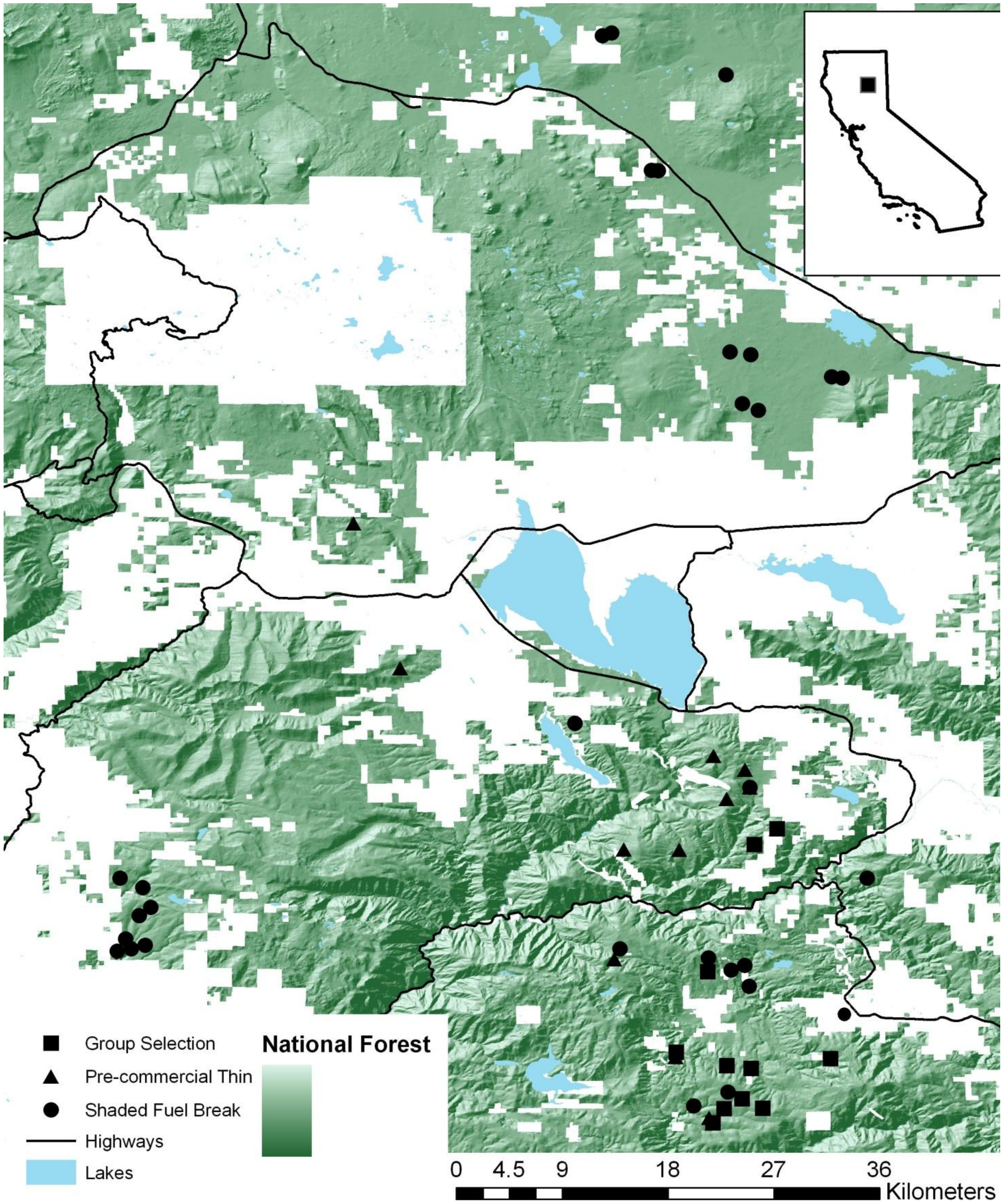

Figure 1. Study area with location of bird surveys (multiple survey points are represented by each marker) in the Plumas and Lassen National Forest, California, USA.

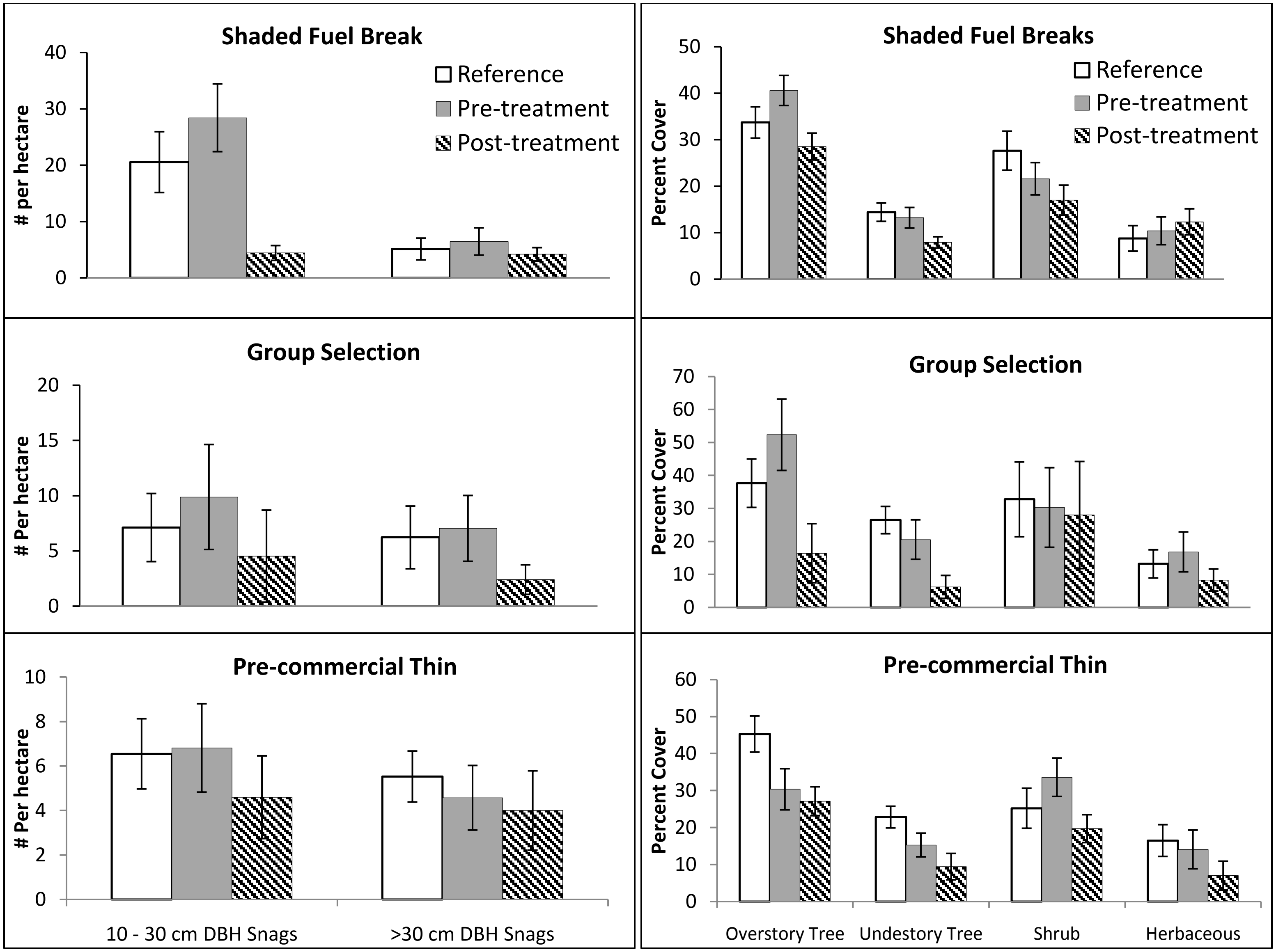

Figure 2. Vegetation characteristics at reference (ref), pre-treatment (pre), and post (treat) fuel treatment avian point count stations in the Plumas and Lassen National Forests, California, USA.

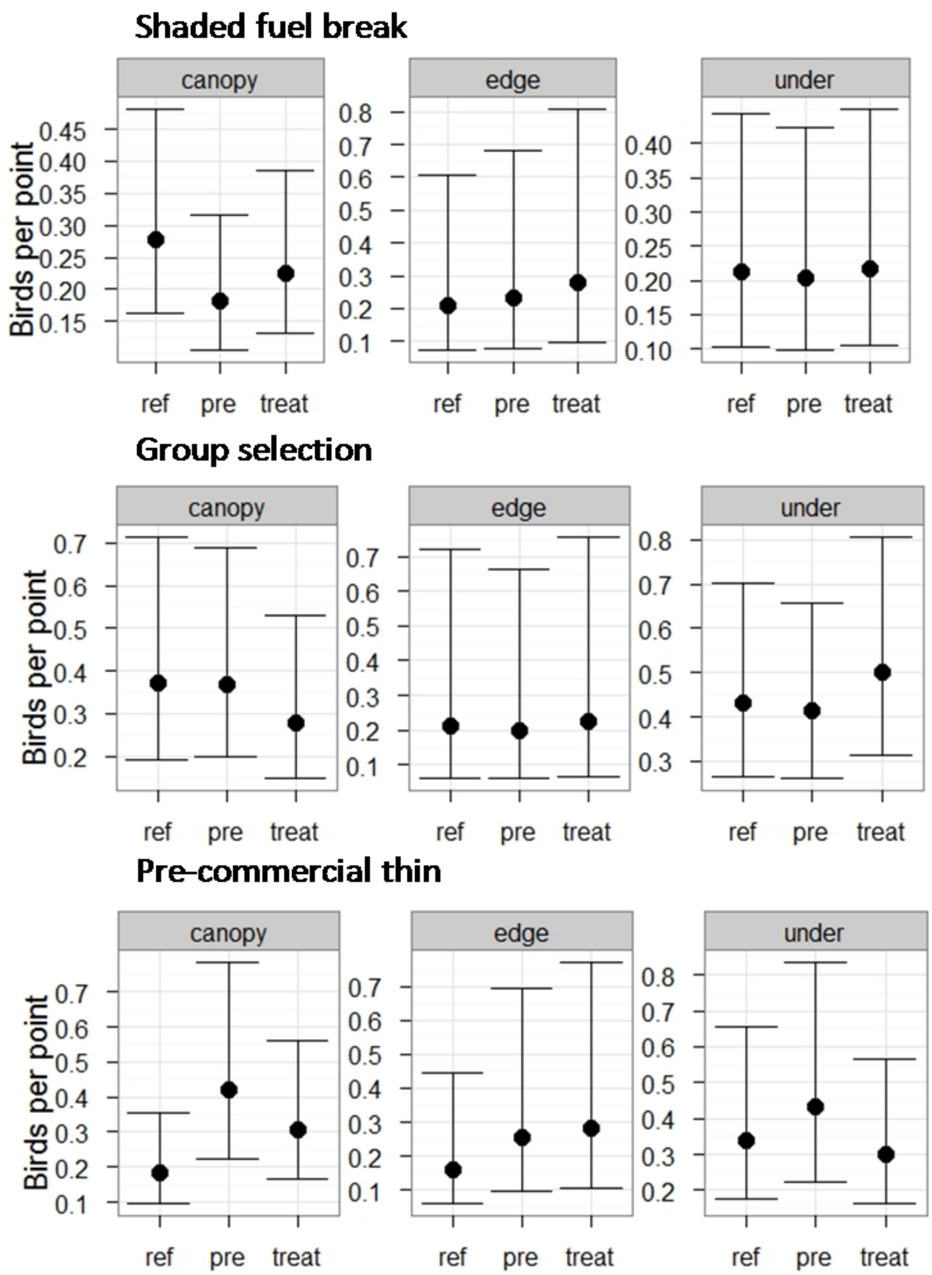

Figure 3. Model averaged per visit abundance (detections <50m from observers) of species in each of three general habitat guilds at reference (ref), pre-treatment (pre), and post-treatment (treat) fuel treatment sites in the Northern Sierra Nevada, California, USA.

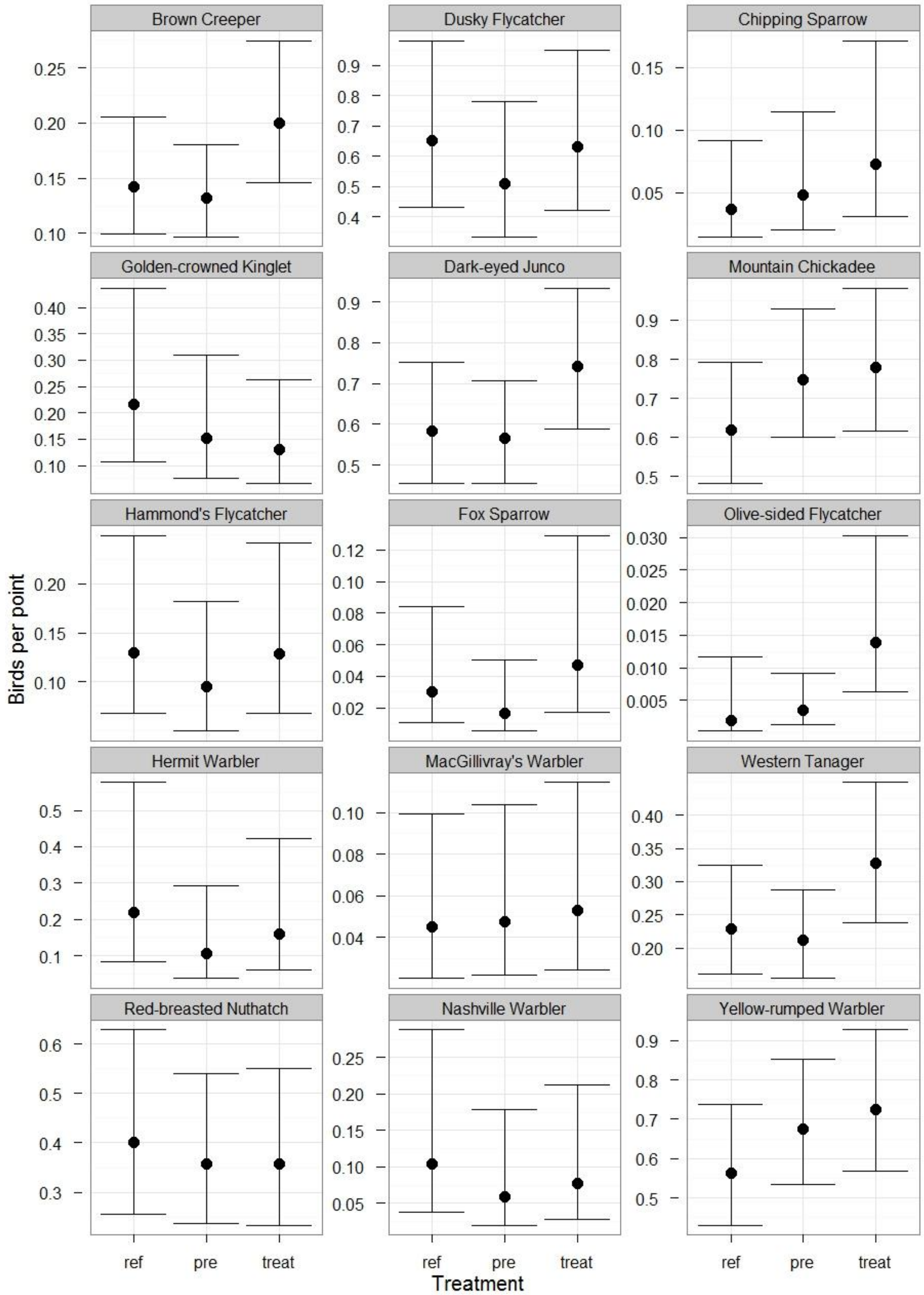

Figure 4. An index of the abundance of 15 passerine bird species at 122 untreated (ref) point count stations and 122 prior to (pre) and following (treat) shaded fuel break treatment in the Northern Sierra Nevada, California, USA. Means are model averaged per visit estimates based on detections <50m from observers. Y-axis scale varies between species.

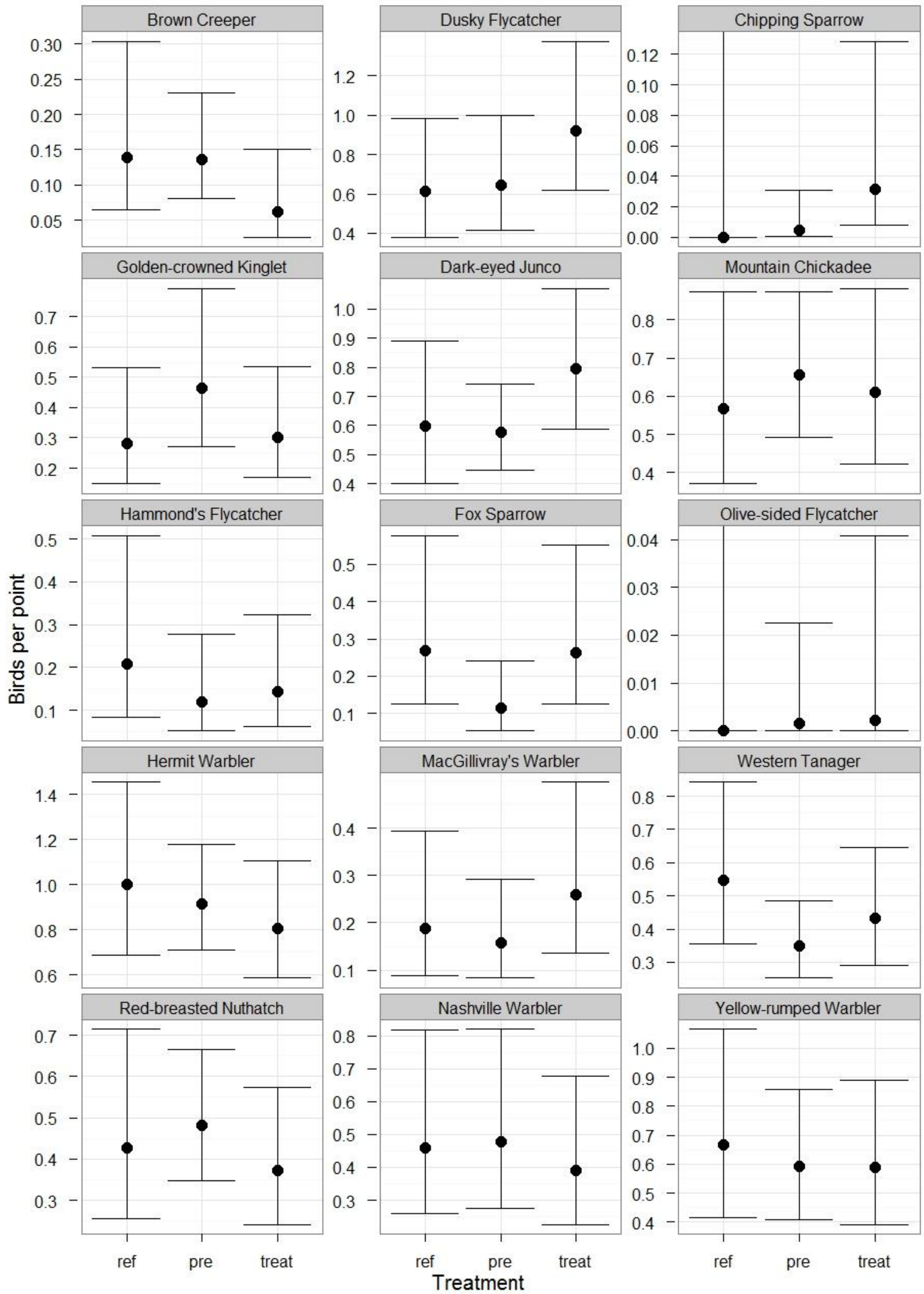

Figure 5. An index of the abundance of 15 passerine bird species at 17 untreated (ref) point count stations and 17 prior to (pre) and following (treat) group selection treatments in the Northern Sierra Nevada, California, USA. Means are model averaged per-visit estimates based on detections <50m from observers. Due to small sample size we were unable to produce meaningful estimates for Olive-sided Flycatcher. Y-axis scale varies between species.

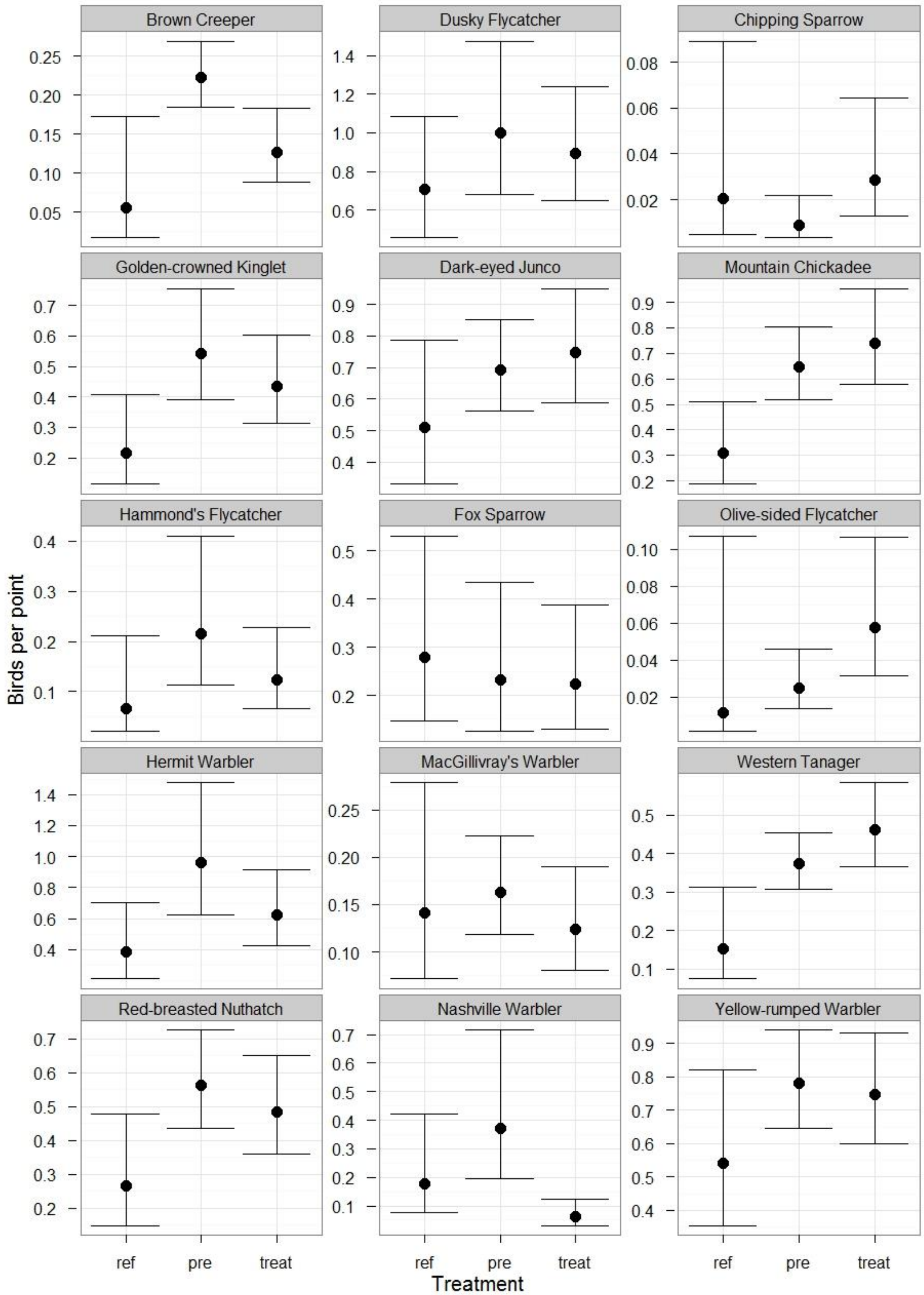

Figure 6. An index of the abundance of 15 passerine bird species at 49 untreated (ref) point count stations and 49 stations prior to (pre) and following (treat) pre-commercial understory thinning treatments in the Northern Sierra Nevada, California, USA. Means are model averaged per visit estimates based on detections <50m from observers. Y-axis scale varies between species.

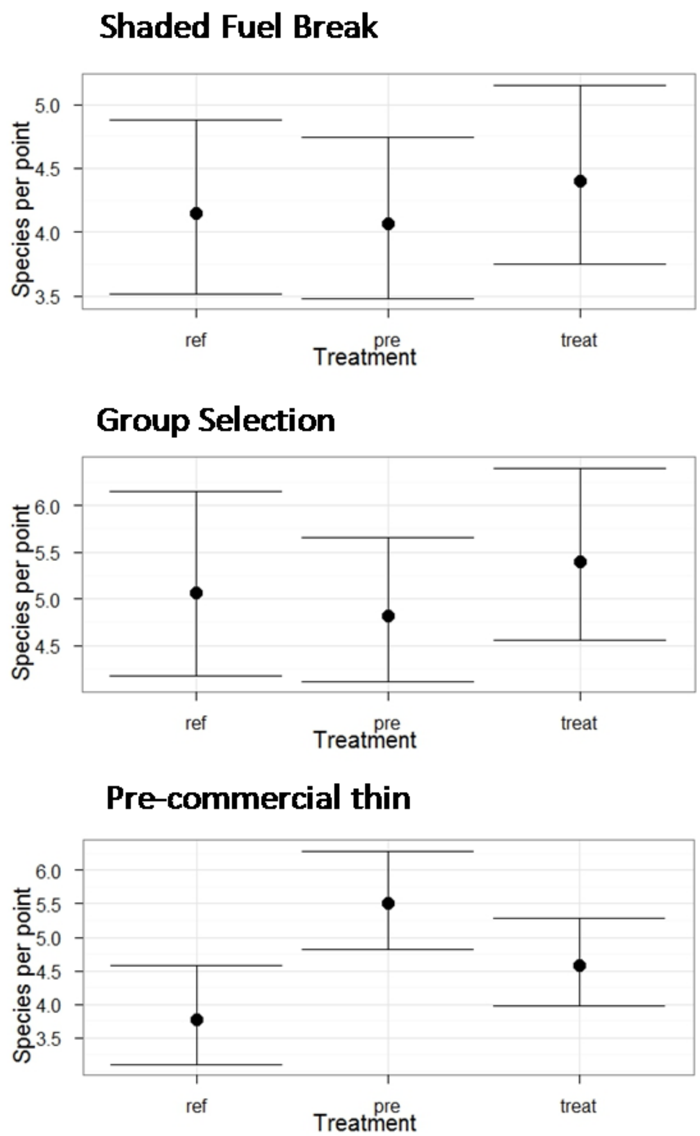

Figure 7. The predicted mean number of avian species detected within 50m of observers per point count station within reference (ref), pre-treatment (pre), and following treatment (treat) sites for three fuels reduction treatment types across the Plumas and Lassen National Forest, California, USA.

Table A.1 Estimates of probability of detection, standard errors and tests of significance obtained separately for each type of fuel reduction treatment (shaded fuel breaks, group selection, and pre-commercial understory thinning) and bird species. Estimates are from a zero-inflation model with detection function that included treatment categories (reference, pre-treatment and treated) as a factor, plus an intercept parameter. The reference category is pre-treatment, therefore estimates with sufficiently low p-value would evidence lack of support for the hypothesis of no difference in detection between treated and untreated samples.

Shaded Fuel Break

Species	Parameter	Estimate	Std. Error	z value	Pr(>|z|)
Yellow-rumped Warbler	Intercept	-1.564	1.196	-1.307	0.191
	Treatment-reference	-0.208	1.355	-0.153	0.878
	Treatment-treated	-8.416	45.209	-0.186	0.852
Brown Creeper	Intercept	1.253	0.586	2.137	0.033
	Treatment-reference	-2.385	1.856	-1.285	0.199
	Treatment-treated	-1.627	1.101	-1.478	0.139
Chipping Sparrow	Intercept	0.649	1.458	0.445	0.656
	Treatment-reference	1.216	1.495	0.813	0.416
	Treatment-treated	0.163	1.654	0.099	0.921
Dark-eyed Junco	Intercept	-0.094	0.346	-0.273	0.785
	Treatment-reference	0.015	0.41	0.037	0.97
	Treatment-treated	0.103	0.414	0.249	0.804
Dusky Flycatcher	Intercept	-0.565	0.304	-1.858	0.063
	Treatment-reference	-0.378	0.408	-0.927	0.354
	Treatment-treated	-0.258	0.481	-0.535	0.593
Fox Sparrow	Intercept	0.932	0.64	1.457	0.145
	Treatment-reference	0.33	0.669	0.492	0.623
	Treatment-treated	0.678	0.696	0.973	0.331
Golden-crowned Kinglet	Intercept	0.579	0.44	1.315	0.189
	Treatment-reference	-0.708	0.53	-1.336	0.181
	Treatment-treated	-10.669	83.672	-0.128	0.899
Hammond’s Flycatcher	Intercept	0.598	0.493	1.212	0.225
	Treatment-reference	0.427	0.56	0.762	0.446
	Treatment-treated	-0.058	0.708	-0.082	0.935
Hermit Warbler	Intercept	0.208	0.288	0.722	0.47
	Treatment-reference	-0.049	0.383	-0.129	0.897
	Treatment-treated	0.356	0.366	0.972	0.331
MacGillivray’s Warbler	Intercept	1.713	0.621	2.757	0.006
	Treatment-reference	-0.798	0.732	-1.089	0.276
	Treatment-treated	0.405	0.856	0.473	0.636

Mountain Chickadee	Intercept	-0.265	0.305	-0.869	0.385
	Treatment-reference	-0.105	0.361	-0.291	0.771
	Treatment-treated	-2.94	3.409	-0.862	0.389
Nashville Warbler	Intercept	0.249	0.42	0.594	0.553
	Treatment-reference	-0.307	0.528	-0.581	0.561
	Treatment-treated	0.61	0.582	1.047	0.295
Olive-sided Flycatcher	Intercept	-4.322	217.49	-0.02	0.984
	Treatment-reference	-3.261	429.087	-0.008	0.994
	Treatment-treated	-5.713	439.45	-0.013	0.99
Red-breasted Nuthatch	Intercept	0.035	0.559	0.062	0.95
	Treatment-reference	0.334	0.594	0.562	0.574
	Treatment-treated	-1.02	0.954	-1.07	0.285
Western Tanager	Intercept	0.496	0.432	1.149	0.251
	Treatment-reference	-0.121	0.571	-0.212	0.832
	Treatment-treated	-0.417	0.589	-0.708	0.479

Group Selection

Species	Parameter	Estimate	Std. Error	z value	Pr(>|z|)
Yellow-rumped Warbler	Intercept	-2.046	4.472	-0.458	0.647
	Treatment-reference	1.233	4.513	0.273	0.785
	Treatment-treated	-9.043	174.029	-0.052	0.959
Brown Creeper	Intercept	-7.826	83.506	-0.094	0.925
	Treatment-reference	7.284	83.547	0.087	0.931
	Treatment-treated	-2.207	327.087	-0.007	0.995
Chipping Sparrow	Not able to estimate				
Dark-eyed Junco	Intercept	-2.855	11.65	-0.245	0.806
	Treatment-reference	0.935	11.853	0.079	0.937
	Treatment-treated	1.753	11.683	0.15	0.881
Dusky Flycatcher	Intercept	-0.696	0.876	-0.794	0.427
	Treatment-reference	-2.389	4.643	-0.514	0.607
	Treatment-treated	-8.018	86.555	-0.093	0.926
Fox Sparrow	Intercept	0.419	0.536	0.782	0.434
	Treatment-reference	-3.208	11.535	-0.278	0.781
	Treatment-treated	0.184	0.767	0.24	0.811
Golden-crowned Kinglet	Intercept	-1.847	6.866	-0.269	0.788
	Treatment-reference	-3.706	55.577	-0.067	0.947
	Treatment-treated	-0.374	9.577	-0.039	0.969
Hammond’s Flycatcher	Intercept	-8.259	67.841	-0.122	0.903
	Treatment-reference	8.898	67.842	0.131	0.896
	Treatment-treated	8.944	67.845	0.132	0.895
Hermit Warbler	Intercept	-9.485	72.15	-0.131	0.895

	Treatment-reference	-2.556	456.744	-0.006	0.996
	Treatment-treated	7.798	72.16	0.108	0.914
MacGillivray's Warbler	Intercept	0.04	1.237	0.032	0.974
	Treatment-reference	0.556	1.328	0.419	0.676
	Treatment-treated	-8.904	86.044	-0.103	0.918
Mountain Chickadee	Intercept	-0.453	0.98	-0.462	0.644
	Treatment-reference	-0.093	1.094	-0.085	0.932
	Treatment-treated	-0.93	1.893	-0.491	0.623
Nashville Warbler	Intercept	-0.161	0.569	-0.283	0.777
	Treatment-reference	-1.808	2.483	-0.728	0.467
	Treatment-treated	-10.025	122.93	-0.082	0.935
Olive-sided Flycatcher	Not able to estimate				
Red-breasted Nuthatch	Intercept	-2.874	11.974	-0.24	0.81
	Treatment-reference	2.699	11.984	0.225	0.822
	Treatment-treated	1.231	12.286	0.1	0.92
Western Tanager	Intercept	-0.79	1.629	-0.485	0.628
	Treatment-reference	0.79	1.762	0.448	0.654
	Treatment-treated	-0.59	3.068	-0.192	0.848

Pre-commercial Thin

Species	Parameter	Estimate	Std. Error	z value	Pr(>|z|)
Yellow-rumped Warbler	Intercept	0.25	0.898	0.279	0.781
	Treatment-reference	-3.299	2.59	-1.274	0.203
	Treatment-treated	-1.882	1.406	-1.339	0.181
Brown Creeper	Intercept	-8.414	231.077	-0.036	0.971
	Treatment-reference	-2.879	537.826	-0.005	0.996
	Treatment-treated	-3.922	570.489	-0.007	0.995
Chipping Sparrow	Intercept	-6.114	219.533	-0.028	0.978
	Treatment-reference	-1.92	485.796	-0.004	0.997
	Treatment-treated	-2.715	490.258	-0.006	0.996
Dark-eyed Junco	Intercept	0.764	0.57	1.341	0.18
	Treatment-reference	-1.321	0.656	-2.013	0.044
	Treatment-treated	-1.967	0.913	-2.155	0.031
Dusky Flycatcher	Intercept	-7.958	33.549	-0.237	0.813
	Treatment-reference	-3.751	112.609	-0.033	0.973
	Treatment-treated	7.31	33.551	0.218	0.828
Fox Sparrow	Intercept	-0.034	0.602	-0.056	0.956
	Treatment-reference	0.301	0.692	0.436	0.663
	Treatment-treated	1.101	0.667	1.651	0.099
Golden-crowned Kinglet	Intercept	-8.938	95.37	-0.094	0.925
	Treatment-reference	-3.965	189.071	-0.021	0.983

	Treatment-treated	-2.304	151.911	-0.015	0.988
Hammond’s Flycatcher	Intercept	-6.398	125.76	-0.051	0.959
	Treatment-reference	6.859	125.76	0.055	0.957
	Treatment-treated	7.383	125.761	0.059	0.953
Hermit Warbler	Intercept	-0.389	1.249	-0.311	0.756
	Treatment-reference	-10.925	47.901	-0.228	0.82
	Treatment-treated	-0.608	1.351	-0.45	0.652
MacGillivray’s Warbler	Intercept	-7.845	71.82	-0.109	0.913
	Treatment-reference	7.479	71.826	0.104	0.917
	Treatment-treated	-0.833	438.969	-0.002	0.998
Mountain Chickadee	Intercept	0.309	0.749	0.412	0.68
	Treatment-reference	-0.663	0.791	-0.838	0.402
	Treatment-treated	-0.413	0.801	-0.515	0.606
Nashville Warbler	Intercept	0.41	1.045	0.392	0.695
	Treatment-reference	-0.701	1.139	-0.615	0.538
	Treatment-treated	1.217	1.531	0.795	0.427
Olive-sided Flycatcher	Intercept	-5.672	176.055	-0.032	0.974
	Treatment-reference	-3.666	309.356	-0.012	0.991
	Treatment-treated	-3.751	314.285	-0.012	0.99
Red-breasted Nuthatch	Intercept	1.128	0.828	1.363	0.173
	Treatment-reference	-1.827	0.938	-1.948	0.051
	Treatment-treated	-9.371	64.136	-0.146	0.884
Western Tanager	Intercept	0.869	1.288	0.674	0.5
	Treatment-reference	-10.447	65.612	-0.159	0.873
	Treatment-treated	-1.699	1.561	-1.088	0.276