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Plumas-Lassen Administrative Study 2010 Post-fire Avian Monitoring Report



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Executive Summary

In 2009 the avian module of the Plumas-Lassen Area Study (PLAS) expanded to address important questions related to post-fire habitat and its management. The primary objective of this new part of the study is to assess the influence of post-fire conditions on spatial and temporal variation in bird abundance, and to use this information to guide forest management actions that can maintain avian diversity across multiple spatial scales. In 2010 we continued sampling three areas affected by fire within the boundaries of the original PLAS study: the Storrie, Moonlight, and Cub fires and 40 transects in the adjacent PLAS green forest treatment units.

We compared the abundance of 30 species that breed in the study area - representing a range of habitat types and conditions - between the three fire areas and green forest study sites in the PLAS. When data from all fires were combined, 10 species were significantly more abundant in fire areas and 11 were significantly more abundant in green forest.

Shannon index of diversity, species richness, and total bird abundance were significantly higher in PLAS green forest in 2010 than any of the three fire areas. Among the three fires, Storrie had the highest indices for each of these three metrics. All three metrics were significantly lower in both the Moonlight and Storrie private lands than in the adjoining Forest Service lands. All three indices were markedly higher in the Moonlight, Storrie, and PLAS in 2010 compared to 2009 though indices were similar in both years for the Cub fire.

Burn severity as measured by the Composite Burn Index was a significant predictor of species richness and total bird abundance in the Moonlight fire, and the effect was curvilinear with highest values at moderate burn index values. Burn severity was not a significant predictor of species richness or total bird abundance in either the Cub or Storrie Fires; in the Storrie Fire the highest values for these metrics were observed in the high severity areas in contrast to the two younger fires.

We found and confirmed 180 active nests of 11 cavity nesting species in 2010. Of those 35 were in Cub, 111 in Moonlight, and 34 in Storrie. We found a total of 54 Hairy Woodpecker nests and 16 Black-backed Woodpecker nests in 2010 –both Management Indicator Species for Sierra Nevada National Forests. The nest trees selected were quite variable in both size and

species. Most woodpecker species did not show preference for nest tree size but nest tree species did vary by bird species. The strongest selection was for decay class and top condition with almost all species selecting for broken top trees with greater decay than those available on the landscape.

Here we present 24 management recommendations; a culmination of our results, scientific literature, and expert opinion. Many of these should be considered hypotheses that we strongly recommend be tested to ensure the best possible management practices are being employed to sustain avian communities in the Sierra Nevada. A list of all outreach associated with this project (including presentations and publications) is available in Appendix A.

Post Fire Habitat Management Recommendations

General

- Whenever possible restrict activities that depredate breeding bird nests and young to the non-breeding season (August - April)
- Consider post-fire habitat as important component of the ecosystem necessary for maintaining biological diversity in the Sierra Nevada
- Consider the area of a fire that burned in high severity, as opposed to the area of the entire fire, when determining what percentage of the fire area to salvage log
- Consider the landscape context (watershed, ranger district, forest, ecosystem) and availability of different habitat types when planning post-fire management actions
- Consider that snags in post-fire habitat are still being used by a diverse and abundant avian community well beyond the 5 to 10 year horizon often suggested

Snags

- Manage a substantial portion of post-fire areas for large patches (minimum of 50 acres) of high severity habitat
- Retain high severity areas in locations with higher densities of larger diameter trees
- Retain high severity patches in areas where pre-fire snags are abundant as these are the trees most readily used in the first five years after a fire

- Retain snags in salvaged areas greater than green forest standards and retain some in dense clumps
- Snag retention immediately following a fire should aim to achieve a range of snag conditions from heavily decayed to recently dead in order to ensure a longer lasting source of snags for nesting birds
- When reducing snags in areas more than five years post fire (e.g. Storrie Fire) snag retention should favor large pine and Douglas Fir but decayed snags of all species with broken tops should be retained in recently burned areas
- Retain snags (especially large pine trees that decay slowly) in areas being replanted as they can provide the only source of snags in those forest patches for decades to come
- Consider retaining smaller snags in heavily salvaged areas to increase snag densities as a full size range of snags are used by a number of species from as little as 6 inches diameter, though most cavity nests were in snags over 16 inches

Early Successional Habitat

- Manage post-fire areas for diverse and abundant understory plant community including shrubs, grasses, and forbs. Understory plant communities provide a unique and important resource for a number of species in a conifer dominated ecosystem
- Most shrub patches should be at least 10 acres and shrub cover should average over 50% across the area in order to support area sensitive species such as Fox Sparrow
- Retain natural oak regeneration with multiple stems (avoid thinning clumps) as these dense clumps create valuable understory bird habitat in post-fire areas 10 – 15 years after the fire
- Manage for a mosaic when treating shrub habitats to ensure some dense patches are retained. In highly decadent shrub habitat consider burning or masticating half the area (in patches) in one year and burning the rest in the following years once fuel loads have been reduced.

- Maximize the use of prescribed fire to create and maintain chaparral habitat and consider a natural fire regime interval of 20 - 30 years as the targeted re-entry rotation for creating disturbance in these habitat types.

Shaping Future Forest

- Limit replanting of dense stands of conifers in areas with significant oak regeneration and when replanting these areas use conifer plantings in clumps to enhance the future habitat mosaic of a healthy mixed conifer hardwood or pine-hardwood stand
- Retain patches of high severity fire adjacent to intact green forest patches as the juxtaposition of unlike habitats is positively correlated with a number of avian species including declining species such as Olive-sided Flycatcher, Western Wood-Pewee, and Chipping Sparrow
- Incorporate fine scale heterogeneity in replanting by clumping trees with unplanted areas interspersed to create fine scale mosaics and invigorate understory plant communities and natural recruitment of shade intolerant tree species
- Plant a diversity of tree species where appropriate as mixed conifer stands generally support greater avian diversity than single species dominated stands in the Sierra Nevada
- Consider staggering plantings across decades and leaving areas to naturally regenerate in order to promote uneven-aged habitat mosaics at the landscape scale
- Consider fuels treatments to ensure the fire resiliency of remnant stands of green forest within the fire perimeter as these areas increase avian diversity within the fire and the edges between unlike habitats support a number of species (e.g. Olive-sided Flycatcher)

Introduction

The primary objective of the landbird module of the Plumas-Lassen Administrative Study is to assess the impact of forest management practices in sustaining a long-term ecologically stable forest ecosystem at the local and landscape scales. We know the avian community in the Sierra Nevada is comprised of species that are associated with a wide range of forest seral stages, vegetative composition, and structures. This vegetation, and hence avian diversity, is constantly changing as a result of natural disturbances (primarily fire) and forest management activities.. Therefore, it is imperative for managers to consider how natural disturbance events interact temporally and spatially with management actions, and how ecological integrity can be achieved in an inherently dynamic system.

In the Sierra Nevada, there is a pressing need to understand the nexus of silvicultural practices, wildfire, and fuels treatments in order to maintain forested ecosystems that are ecologically diverse and resilient. In the context of a century of fire suppression, at the core of the debate over how to manage Sierra forests is how to most appropriately manage areas where natural disturbance regimes have been disrupted. Land managers need more information about the suitability of habitat created through fire suppression, fuel treatments (DFPZ, groups, mastication), and wildfire and post-wildfire management to ensure the goals of maintaining biological diversity are achieved.

The challenge of integrating wildfire and forest management into wildlife conservation is not unique to the Sierra Nevada. Because large, infrequent disturbances are responsible for long-lasting changes in forest structure and composition (Foster et al. 1998), they are recognized as a critical element of bird community dynamics (Brawn et al. 2001). In many regions of western North America, fires burn with considerable spatial and temporal variability (Agee 1993), creating complex mosaics of vegetation patches. In these systems, changes in bird abundance are often linked to post-fire vegetation characteristics and landscape composition (Saab et al. 2002, Huff et al. 2005, Smucker et al. 2005).

In addition to fire suppression, there are a number of management activities that influence post-fire vegetation characteristics and landscape composition in working forests. These activities include salvage-logging, the mechanical mastication and herbicidal treatments

to reduce broadleaf shrubs, and planting of conifer species that are favored by forestry. As a result, management activities may have profound influences on post-fire conditions- locally and across landscapes.

Beginning in 2009 the avian module of the Plumas-Lassen Administrative Study (PLAS) expanded to address important questions related to post-fire habitat and its management. The primary objective of this part of the study is to assess the influence of post-fire conditions on spatial and temporal variation in bird abundance, and to use this information to guide forest managers toward practices that can maintain avian diversity across multiple spatial scales. We began sampling three areas affected by fire within the boundaries of the original PLAS study: the Storrie Fire that burned in the fall of 2000, the Moonlight Fire that burned in the Fall of 2007, and the Cub Fire that burned in the summer of 2008. Each of these fires burned at similar elevations and through primarily mixed conifer and true fir vegetation communities, but with varying severity patterns. This report provides results from the first two years of fire monitoring and uses ongoing monitoring of unburned actively managed “green” forest in the study area to provide context.

Methods

Study Location

The Plumas-Lassen Area Study avian module study encompasses portions of the Mount Hough Ranger District of Plumas National Forest and the Almanor Ranger District of Lassen National Forest in the Sierra Nevada Mountains of Northeastern California (Figure 1). In 2009 we added three separate burned areas to our study within this same area. The elevations of sites surveyed ranged from 1126 – 1998 m with a mean of 1658 in the Cub fire, 1199 – 2190 m with a mean of 1779 in the Moonlight Fire, 1107 – 2011 m with a mean of 1528 m in the Storrie fire, and 1094 – 1902 m with a mean of 1483 m at the existing PLAS green forest sites.

Site Selection

Random starting points for each fire transect were generated in ArcGIS 9.2 within the boundaries of each fire (ESRI 2004). The original sampling area was limited to forest service

land and sites with a slope of less than 40 degrees to allow access and safe navigation on foot in a timely manner. We maintained a minimum distance between transect starting points of 1500 m to ensure transects would not overlap and maximize a spatial balance within the sampling frame of each fire. Four more points were added to the starting point on a random compass bearing at 250 m spacing resulting in a 1 km long, five-point transect. We minimized the number of point counts to allow ample time to conduct cavity nest searches during the prime morning hours when bird activity is greatest.

Even with stratification to eliminate areas with steep slopes we had to drop approximately 10% of the original transects selected following field reconnaissance due to issues with access, safe navigation, or noise from nearby streams. We replaced the majority of these transects with new locations until nearly all surveyable areas in each fire were covered. However, due to steep topography and large roadless areas in the Storrie fire – and to a lesser degree the Cub fire – our sampling is not evenly distributed across these fires as it is in the Moonlight.

In 2010 we added an additional ten transects on private land managed by W. M. Beaty & Associates, Inc; six in the Moonlight Fire, and four in the Storrie Fire. These transects were selected using the same protocol as described above, but in this case, the sample area was defined as the private land within the fire boundary, shown in white on the map (Figure 1). We then selected the maximum number of transects that the sample area size would allow.

A total of 60 transects (300 stations) were surveyed across the three fires in 2010. A total of 32 transects were surveyed in the Moonlight fire (including 6 on private land), 13 in the Cub Fire, and 15 in the Storrie Fire (including 4 on private land). Snow prevented access to two Storrie transects (ST04 and ST15) before July, thus they were not surveyed in 2010. Site selection for PLAS green forest study sites followed a similar random selection protocol except each transect contained 12 points instead of five, and approximately 25% of transects were systematically established in areas where treatments were planned (many now implemented). The PLAS site selection protocol for the unburned “green forest” sample is described in detail in the original PLAS study plan and previous annual reports (Stine et al. 2005, Burnett et al. 2009).

Figure 1. The location of PRBO study sites in the Plumas-Lassen study area in 2010.

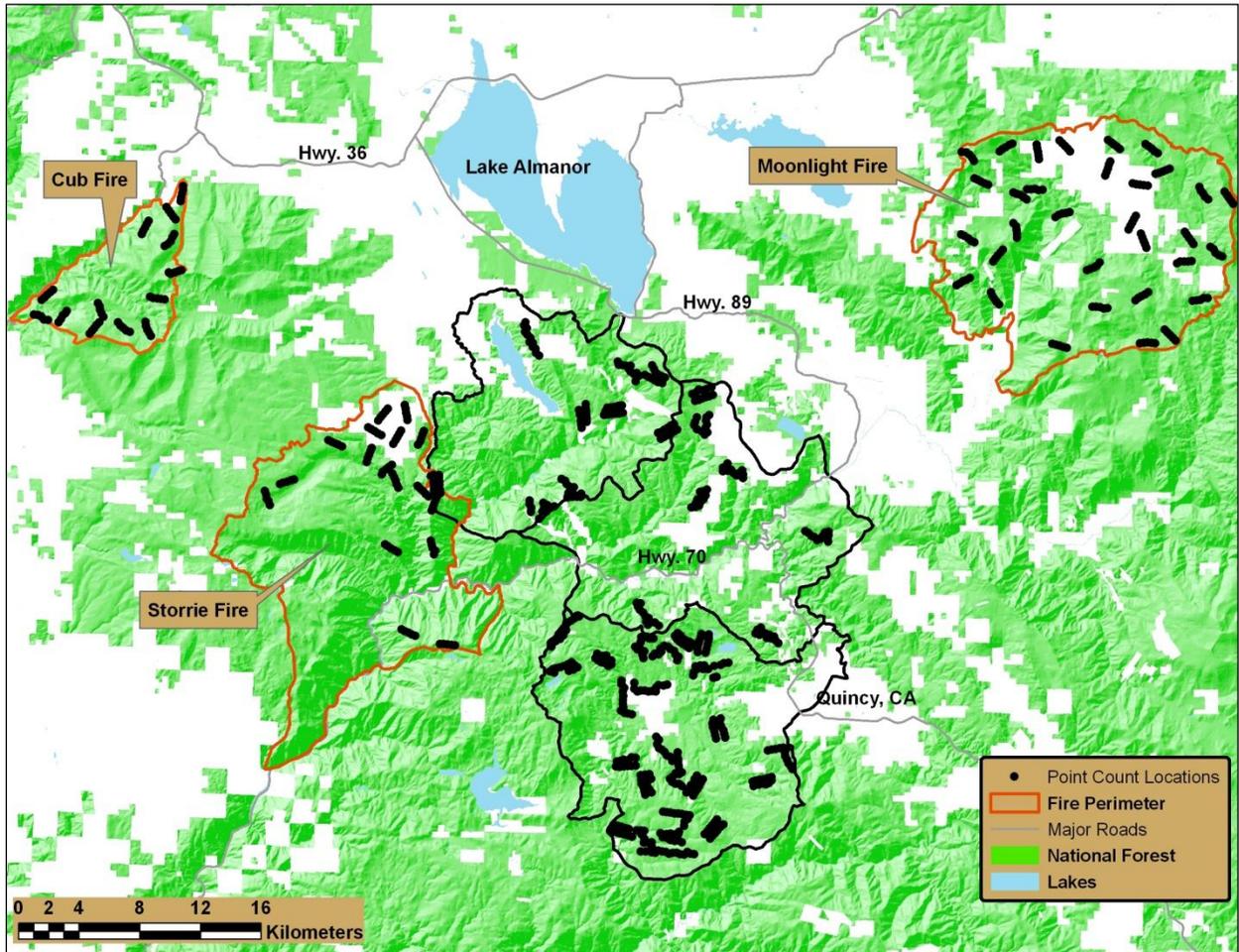
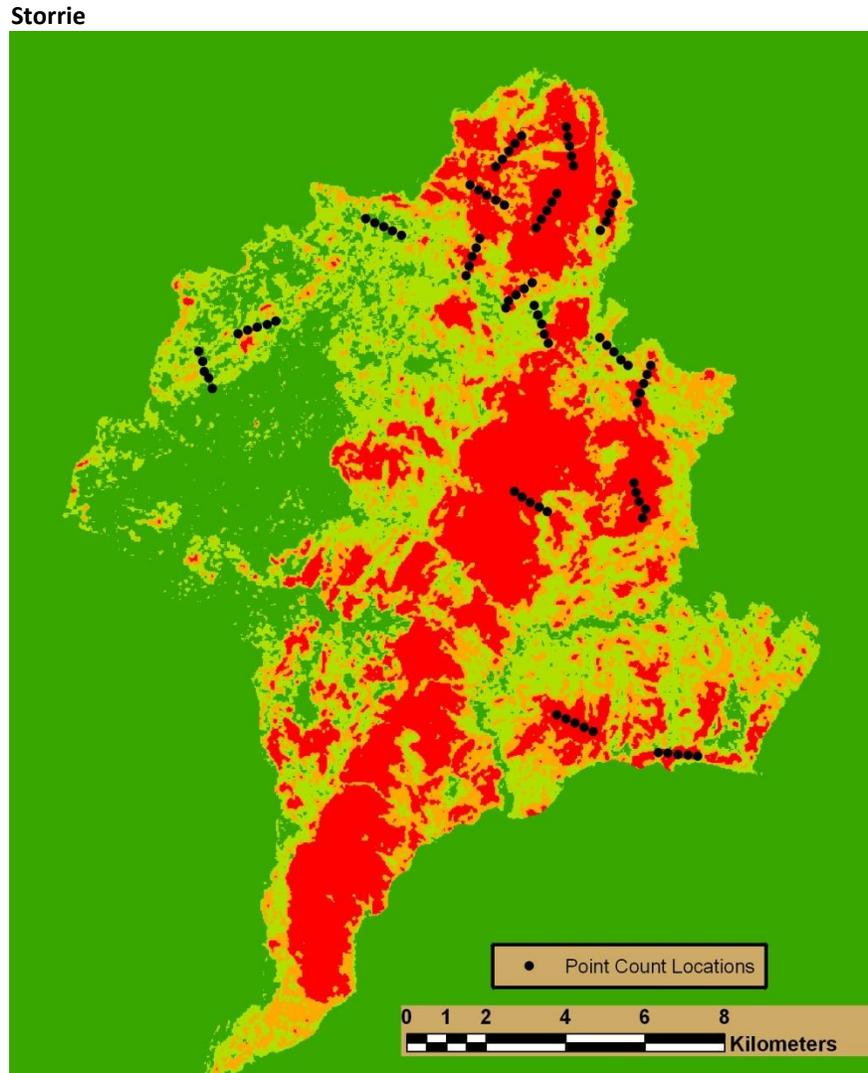
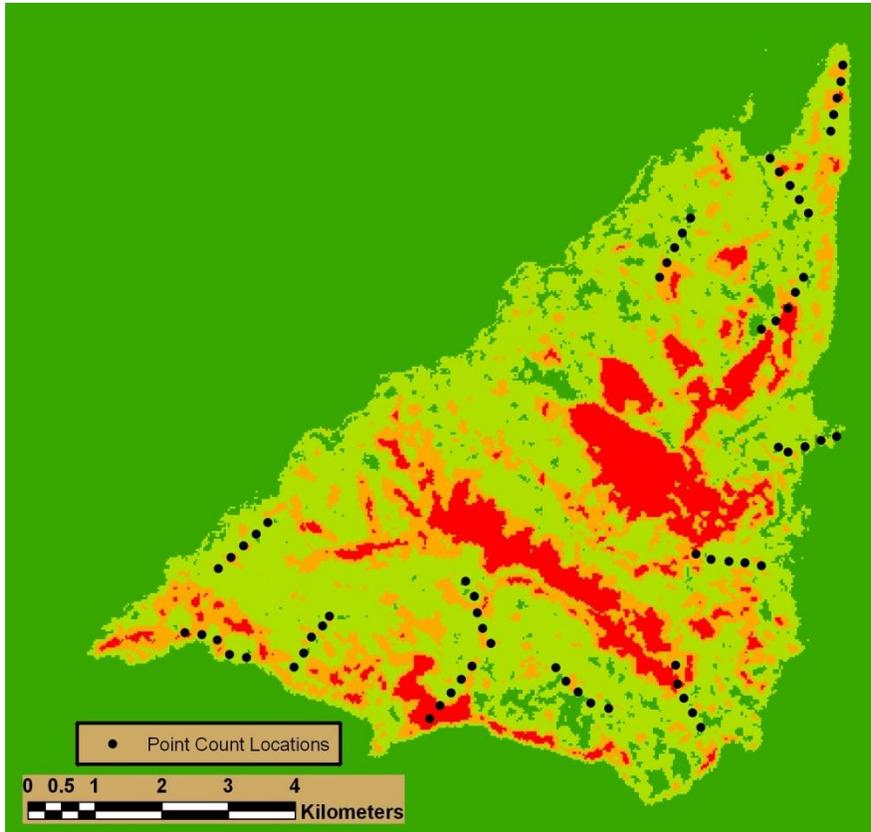


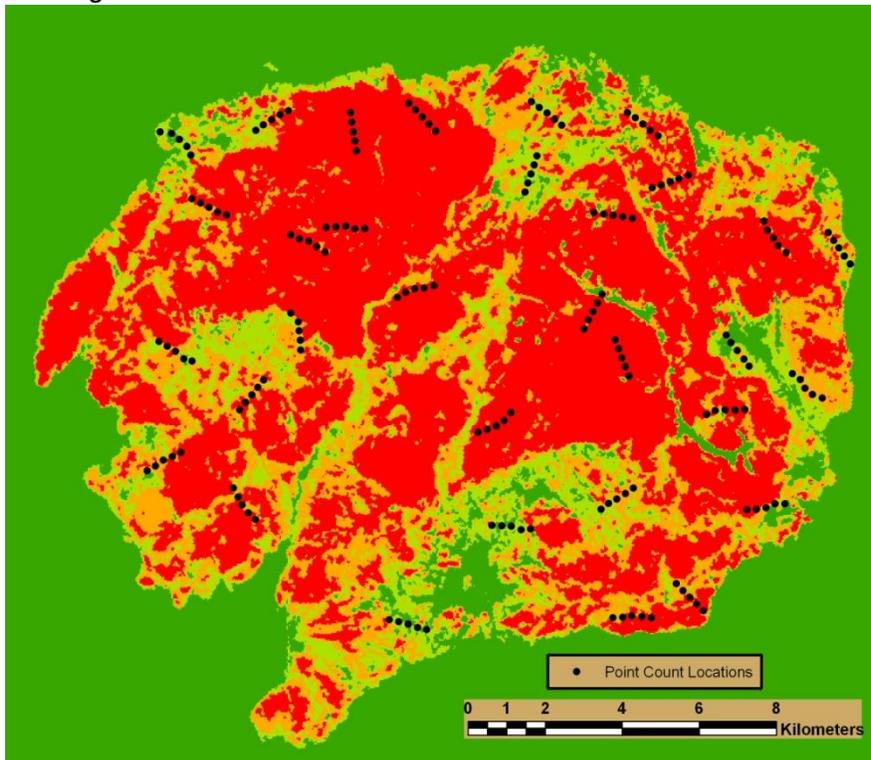
Figure 2. Location of PRBO point count locations overlaid on composite burn index fire severity maps for each of three fires in the study area. Red = high severity, Orange = moderate, Lime = low severity, and green is unburned.



Cub



Moonlight



Bird community surveys

The avian community was sampled using a five minute, exact-distance, point count census (Reynolds et al. 1980, Ralph et al. 2005). 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. The method of initial detection (song, visual, or call) for each individual was also recorded. All observers underwent an intensive, three week training period focused on bird identification and distance estimation prior to conducting surveys. Laser rangefinders were used to assist in distance estimation at every survey point. Counts began around local sunrise, were completed within four hours, and did not occur in inclement weather. Aside from the two transects that were not accessible for the entire season (ST04 and ST15), all but 2 transects were visited twice during the peak of the breeding season from mid May through the first week of July. Cub 12 and Storrie 12 were only visited once due to administrative forest closures in those areas. A list of all sites surveyed in the study area in 2010 and dates surveyed is presented in Appendix B.

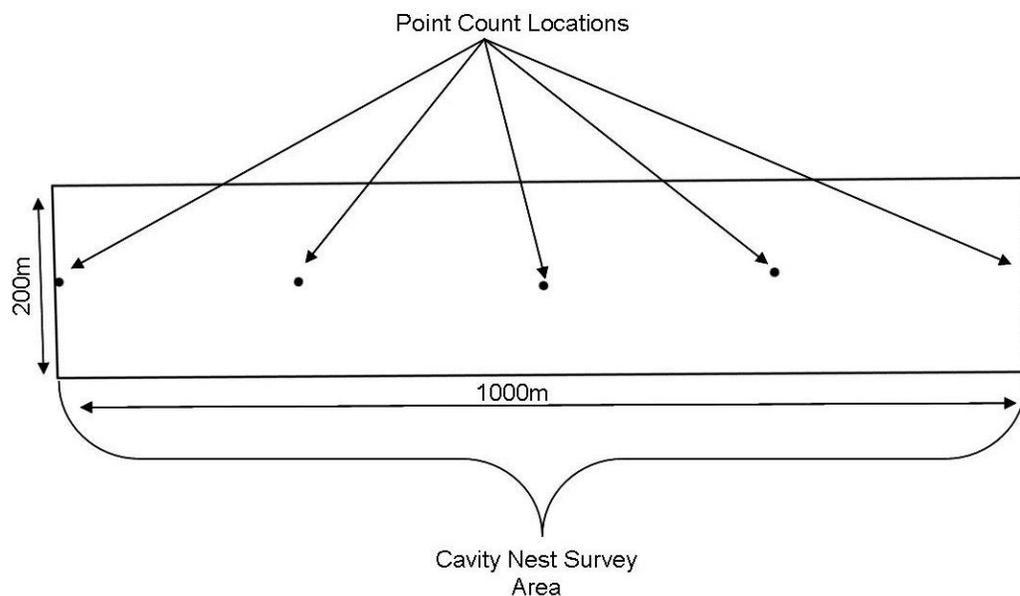
Cavity nest surveys

In addition to point count censuses, at each fire transect a 20 ha area (200 x 1000 m rectangle) surrounding the point count stations was surveyed for nests of cavity-nesting birds following the protocol outlined in "A field protocol to monitor cavity-nesting birds" (Dudley and Saab 2003). In order to focus our attention on species of interest we ignored some of the more common cavity-nesters. Our focal species included both species of bluebird, all woodpeckers, and all cavity-nesting raptors.

After the point count census was complete, the nest survey was conducted for between two and four hours depending on the habitat, terrain and time spent waiting to confirm a cavity's status. All nest surveys were completed by noon. The primary search method for finding nests was bird behavior though once suspicious birds were located observers often conducted a systematic search of snags in the vicinity. Once a potential nest was found, it was observed from a distance for approximately 20 minutes to confirm the cavity was an active nest. If that cavity was confirmed active, a variety of characteristics of both the nest tree and

the cavity were recorded. These characteristics included diameter at breast height (DBH), tree height, tree species, cavity height, tree decay class, and the orientation of the cavity opening. For tree decay, we used a qualitative scale of decay ranging from one to eight, with one being a live, intact tree and eight being a severely decayed stump (Appendix C). If the observer was unable to confirm the cavity was active, its location was recorded to aid nest searchers during the second visit. Only confirmed active nests were used in analysis presented herein.

Figure 3. PRBO Northern Sierra post-fire habitat survey plots.



Vegetation surveys

Vegetation data was collected at all post-fire points in 2010. We measured vegetation characteristics within a 50 m radius plot centered at each point count station following the relevé protocol outlined in the original PLAS bird module study plan (Stine et al. 2005). On these plots we measured shrub cover, live tree cover, and herbaceous cover as well as the relative cover of each species in the shrub and tree layers through ocular estimation. We also collected basal area of live trees and snags using a 10-factor basal area key. To estimate the density of snags across the plot, we recorded data (e.g. DBH, species, height, decay) on every snag within 11.3 m of the center of the point count location. In addition to the point count stations, we collected the same snag data at all active nests, as well as at 5 random locations distributed

throughout the 20 ha nest plot. To select these random points, coordinates were first generated in ArcGIS 9.2 to serve as a guide (ESRI 2004). Once in the field, the observer navigated to within 10 m of the random point and then chose the closest tree over 12 cm DBH as the center of the random snag plot. The center trees of these random snag plots were used as a sample of “random nest trees” and all data collected for active nests were also collected for these random “nests.”

Analysis

A per-point index of abundance (detections within 50 m of observer per visit) at all 798 point count stations was calculated for 30 species that occurred in at least one of the study areas. The species selected were comprised of all of the California Partner’s in Flight Coniferous Forest Focal species (CALPIF 2002), for which we had adequate detections to conduct meaningful analysis, as well as three woodpecker species, and a suite of other species that were among the most abundant in the study area and represented a range of habitat conditions.

In order to quantify the overall songbird community in the study areas we used three different metrics, the Shannon Index of species diversity, species richness, and total bird abundance. The Shannon index used a transformation of Shannon’s diversity index (or H' , Krebs 1989) denoted N_1 (MacArthur 1965). The transformation expresses the data in terms of number of species and thus is more easily interpreted. Expressed mathematically:

$$N_1 = e^{H'} \text{ and } H' = -\sum_{i=1}^{i=S} (p_i)(\ln p_i)$$

Where S = total species richness and p_i is the proportion of the total numbers of individuals for each species (Nur et al. 1999). High Shannon index scores indicate both high species richness and more equal distribution of individuals among species. Species richness is defined simply as the number of species detected within 50m of each point summed across the two visits and total bird abundance is the sum of all species detected per visit within 50 m. All species that do not breed or naturally occur in the study area and those that are not adequately sampled using the point count method including waterfowl, shorebirds, waders, and raptors were excluded

from each calculation. These metrics were investigated for each fire, and the Plumas Lassen Administrative Study green forest study sites (Figure 4).

We compared the abundance of 30 species between all point count stations on Forest Service land across all three fires combined and the PLAS green forest sample using negative binomial regression with species abundance as the independent variable and burn status as the dependent variable using data combined from 2009 and 2010. We dropped the two transects (ST12 and CB12) that were only visited once in 2010 from this analysis. We present mean detections within 50 m of observers per point count station per visit with standard error.

To classify fire severity we used the Composite Burn Index (CBI). This index represents the magnitude of effects caused by the fire and incorporates various strata including changes to soil, amount of vegetation and fuels consumed, re-sprouting from burned plants, establishment of new colonizing species, and blackening or scorching of trees (Key and Benson 2005). The values of CBI were derived from the Relative Differenced Normalized Burn Ratio (RdNBR) which was in turn derived from Landsat Thematic Mapper imagery, a method discussed by Miller and Thode (2007). The Landsat data used to derive CBI was collected on 23 July 2008 in the Cub Fire, 7 July 2008 for the Moonlight Fire, and 21 August 2001 for the Storrie Fire (USDA Forest Service). The values range from zero, representing unburned, to three, representing the highest severity. We conducted the analysis using continuous values between zero and three but for ease of interpretation graphs separate CBI into eight discrete intervals.

We investigated the effect of CBI on species richness and total bird abundance across each of the three fires. We excluded all sites that had been treated (e.g. salvaged or masticated) and all private lands. After investigating the effect of year, and finding consistent patterns across both years in all three fires we combined data from 2009 and 2010 and controlled for year in all models. Thus our sample size for Moonlight was 255 points, for Cub it was 125, and for Storrie it was 118. We used linear regression with untransformed data to evaluate the effect of CBI on these metrics. For each metric we ran the models again with a quadratic term and compared the goodness of fit between the quadratic and linear models using a likelihood ratio test. We ran all tests using Stata statistical software (Stata Corp. 2007). For each best fit model we present F-statistics and p-values and for goodness of fit tests we

present χ^2 likelihood ratio statistics and associated p-values. We assumed statistical significance at the $p < 0.05$ level for all statistical tests.

Results

Avian Community Composition

A total of 101 species were recorded during point count surveys in the study area in 2010. Once species not adequately sampled using the point count method were removed (see methods above), there were 52 species in both the Cub and Storrie Fires and 64 in the Moonlight Fire and PLAS green forest.

We compared the abundance of 30 species that breed in the study area and represent a range of habitat types and conditions between the three fire areas and green forest in the PLAS (Table 1). When data from all fires was combined, ten species (Hairy Woodpecker, Black-backed Woodpecker, White-headed Woodpecker, Western Wood-Pewee, Mountain Bluebird, American Robin, Lazuli Bunting, Chipping Sparrow, Oregon Junco, and Cassin's Finch) were significantly more abundant in fire areas and eleven species (Hammond's Flycatcher, Dusky Flycatcher, Cassin's Vireo, Mountain Chickadee, Red-breasted Nuthatch, Golden-crowned Kinglet, Nashville Warbler, Audubon's Warbler, Hermit Warbler, Black-headed Grosbeak, and Western Tanager) were significantly more abundant in the PLAS green forest. However, despite being less abundant in all fires combined, Mountain Chickadee, Black-headed Grosbeak, and Western Tanager reached their greatest abundance in one of the fires.

The five species with the highest index of abundance (in descending order) in the Cub fire were Mountain Chickadee, Red-breasted Nuthatch, Oregon Junco, Audubon's Warbler, and Hermit Warbler. In the Moonlight they were Oregon Junco, Lazuli Bunting, Dusky Flycatcher, Western Tanager, and Mountain Chickadee. In the Storrie fire they were Fox Sparrow, Oregon Junco, Nashville Warbler, Lazuli Bunting, and Spotted Towhee. In the PLAS green forest study area they were Hermit Warbler, Nashville Warbler, Oregon Junco, Red-breasted Nuthatch, and Dusky Flycatcher. Oregon Junco was the only species among the five most abundant across each of the three fire areas and the PLAS green forest.

Table 1. An index of the abundance of 30 species in each of three burned areas and the adjacent unburned Plumas-Lassen Area Study (PLAS) in 2009 & 2010 combined. Species are listed in taxonomic order and means are presented in bold with standard errors (SE).

	Cub	SE	Moonlight	SE	Storrie	SE	PLAS	SE
Mountain Quail	0.01	0.01	0.01	0.00	0.03	0.01	0.00	0.00
Calliope Hummingbird	0.00	0.00	0.05	0.01	0.05	0.02	0.03	0.00
Hairy Woodpecker	0.10	0.02	0.13	0.02	0.03	0.01	0.02	0.00
White-headed Woodpecker	0.10	0.03	0.05	0.01	0.03	0.01	0.02	0.00
Black-backed Woodpecker	0.02	0.01	0.03	0.01	0.00	0.00	0.00	0.00
Olive-sided Flycatcher	0.02	0.01	0.02	0.01	0.05	0.01	0.02	0.00
Western Wood-Pewee	0.03	0.01	0.06	0.01	0.10	0.03	0.03	0.01
Hammond's Flycatcher	0.15	0.04	0.13	0.02	0.04	0.01	0.21	0.01
Dusky Flycatcher	0.14	0.03	0.33	0.03	0.20	0.03	0.38	0.02
Cassin's Vireo	0.02	0.11	0.02	0.01	0.04	0.01	0.19	0.01
Warbling Vireo	0.08	0.03	0.09	0.02	0.04	0.02	0.09	0.01
Mountain Chickadee	0.39	0.05	0.28	0.03	0.21	0.03	0.38	0.02
Red-breasted Nuthatch	0.37	0.04	0.12	0.02	0.13	0.03	0.39	0.02
Brown Creeper	0.15	0.03	0.11	0.02	0.14	0.04	0.12	0.01
Golden-crowned Kinglet	0.11	0.03	0.07	0.02	0.05	0.02	0.30	0.01
Mountain Bluebird	0.02	0.01	0.08	0.02	0.01	0.01	0.00	0.00
American Robin	0.04	0.01	0.08	0.02	0.09	0.03	0.05	0.01
Nashville Warbler	0.08	0.03	0.06	0.01	0.39	0.06	0.43	0.02
Yellow Warbler	0.03	0.01	0.02	0.01	0.10	0.03	0.05	0.01
Audubon's Warbler	0.33	0.04	0.24	0.02	0.14	0.03	0.35	0.02
Hermit Warbler	0.16	0.03	0.13	0.02	0.09	0.03	0.70	0.02
MacGillivray's Warbler	0.06	0.02	0.13	0.02	0.21	0.04	0.15	0.01
Western Tanager	0.16	0.03	0.29	0.03	0.09	0.03	0.27	0.01
Black-headed Grosbeak	0.05	0.02	0.02	0.01	0.11	0.03	0.09	0.01
Lazuli Bunting	0.02	0.02	0.34	0.04	0.36	0.06	0.03	0.01
Spotted Towhee	0.05	0.02	0.03	0.01	0.28	0.05	0.07	0.01
Chipping Sparrow	0.00	0.00	0.13	0.02	0.13	0.03	0.03	0.00
Fox Sparrow	0.08	0.02	0.25	0.03	0.45	0.07	0.21	0.02
Oregon Junco	0.33	0.04	0.58	0.04	0.39	0.05	0.42	0.02
Cassin's Finch	0.02	0.01	0.06	0.01	0.03	0.02	0.01	0.00

Comparing only Forest Service land, Shannon index of diversity, species richness and total bird abundance were all highest in PLAS green forest in 2010 (Figure 4). In the PLAS Shannon diversity was 6.53, species richness was 7.32, and total bird abundance was 5.95. Among the three fires, Storrie had the highest indices for each of these three metrics with

Shannon diversity of 5.60, species richness of 6.20, and total bird abundance of 5.36. All three metrics were significantly lower in both the Moonlight and Storrie private lands than in the adjoining Forest Service lands. All three indices were also considerably higher in the Moonlight, Storrie, and PLAS in 2010 compared to 2009.

Effect of Burn Severity on Avian Community Metrics

The effects of fire severity on avian community metrics varied by fire in 2010 (Figure 5). Species richness was highest for burn severity between 0.5 and 1.0 in the Moonlight fire, between 1.5 and 2.0 in the Cub fire, and 2.0 – 2.5 in the Storrie Fire. In the Moonlight fire, the overall effect of burn severity on species richness was significant ($F = 11.04$, $p < 0.001$) and negative though a quadratic term significantly improved model fit ($LR\chi^2 = 8.12$, $p = 0.004$). Burn severity had a small, non-significant effect in the Cub fire ($F = 0.01$, $p = 0.99$) and a quadratic term improved model fit but not significantly so ($LR\chi^2 = 1.14$, $p = 0.29$). In the Storrie fire burn severity had a non-significant positive effect on species richness ($F = 1.52$, $p = 0.22$) with little support for a non-linear relationship ($LR\chi^2 = 1.26$, $p = 0.26$). Total bird abundance showed similar patterns across the three fires. The effect of burn severity on total bird abundance in the Moonlight fire was significant ($F = 8.70$, $p < 0.001$), with a quadratic function a better fit than linear ($LR\chi^2 = 5.20$, $p = 0.02$). The effect of burn severity on total bird abundance was not significant for the Storrie fire ($F = 1.10$, $p = 0.35$) and there was little support for a quadratic effect ($LR\chi^2 = 0.07$, $p = 0.79$). The burn severity was not a significant predictor of total bird abundance in the Cub fire either ($F = 0.04$, $p = 0.96$) and although the graphed data shows some signs of a quadratic effect, it was not-significantly better than a linear fit ($LR\chi^2 = 0.28$, $p = 0.59$).

Figure 4. An index of avian species richness, ecological diversity (Shannon-Weaver index), and total bird abundance within 50m of point count stations on Forest Service and private land in the Storrie, Moonlight, and Cub fires in 2010 with 95% confidence intervals.

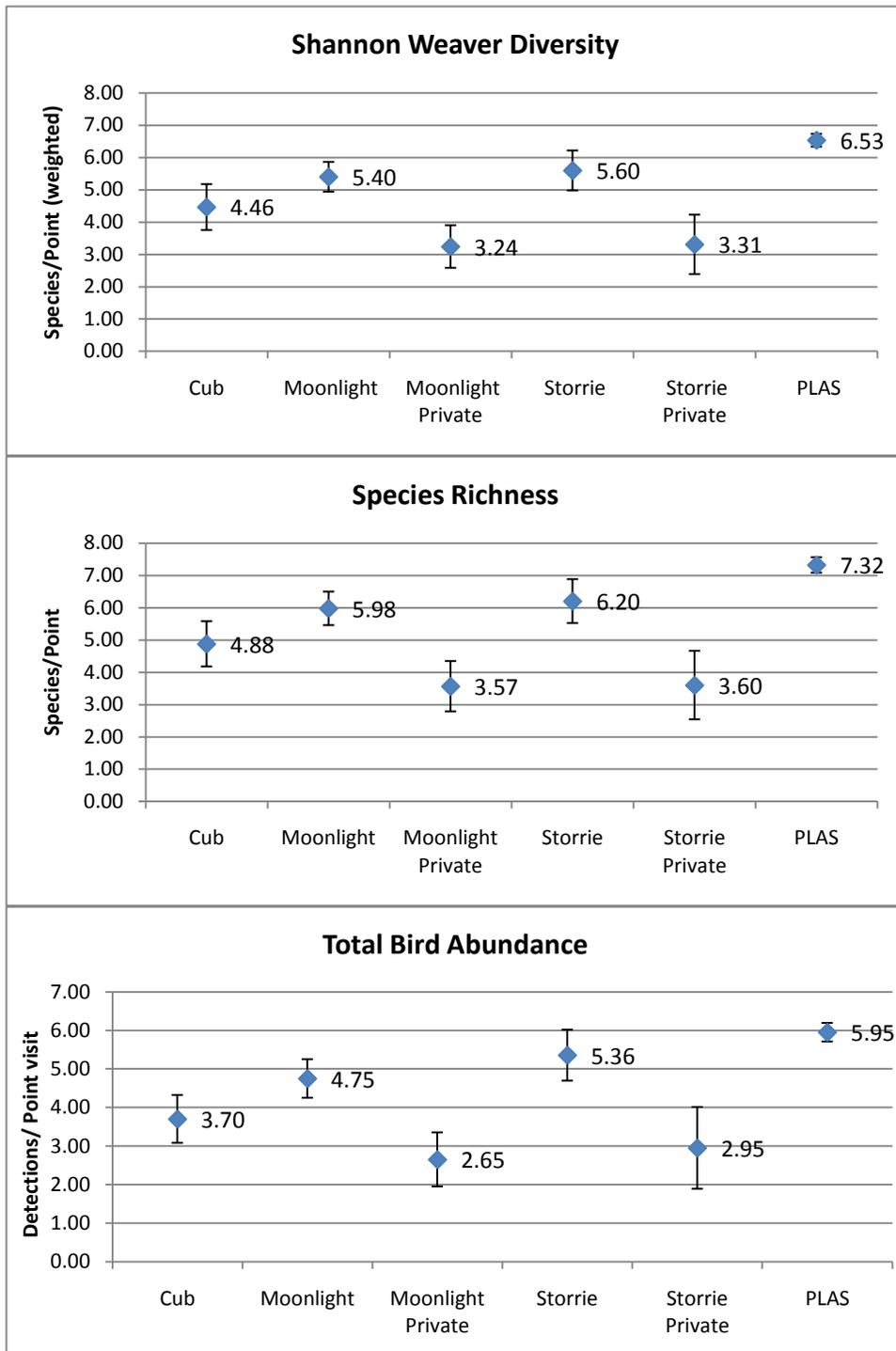


Figure 5. The effect of burn severity (Composite Burn Index) on avian community indices at three fires in 2009 & 2010 with best fit line (based on likelihood ratio test) and 95% confidence intervals.

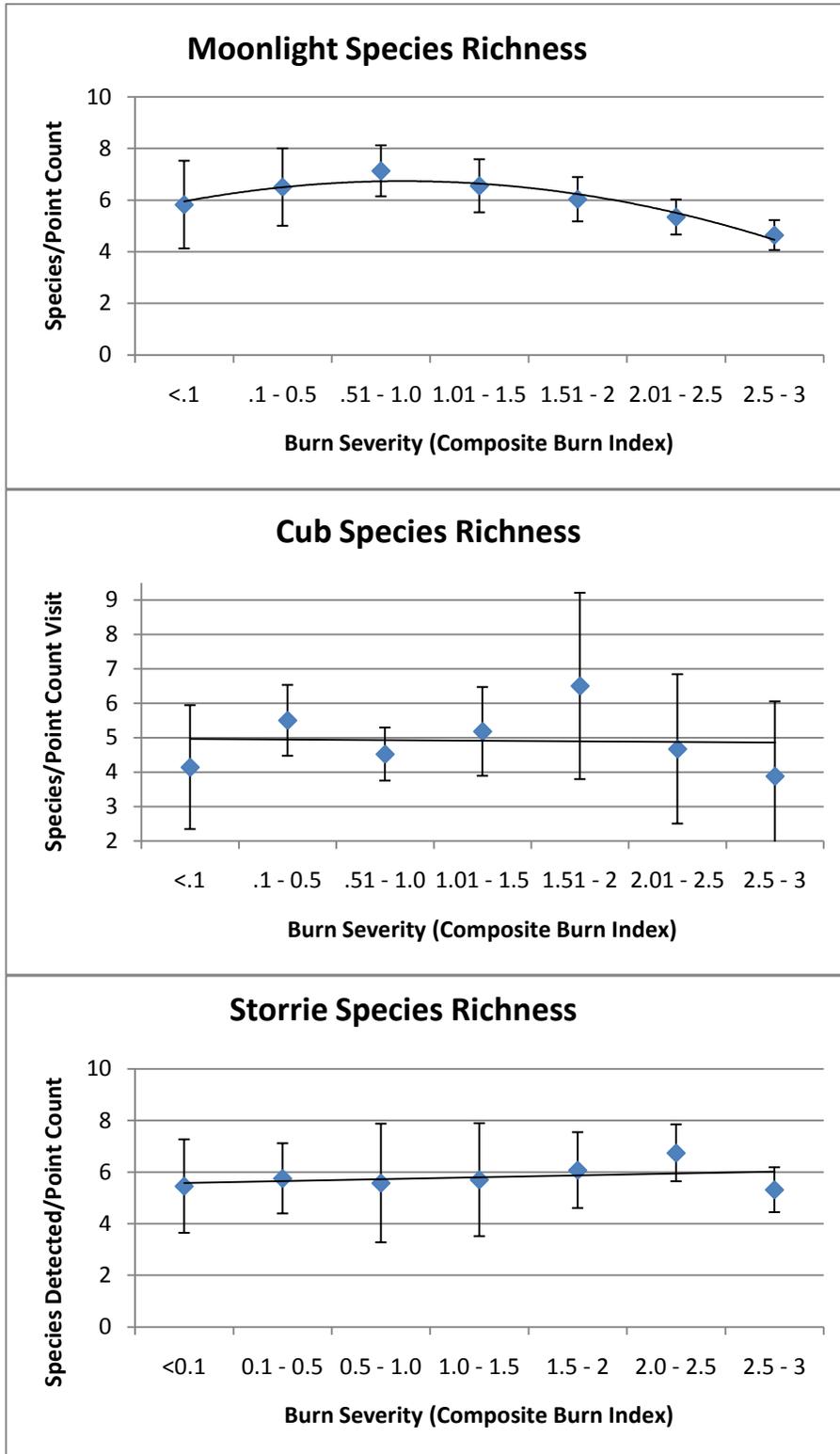
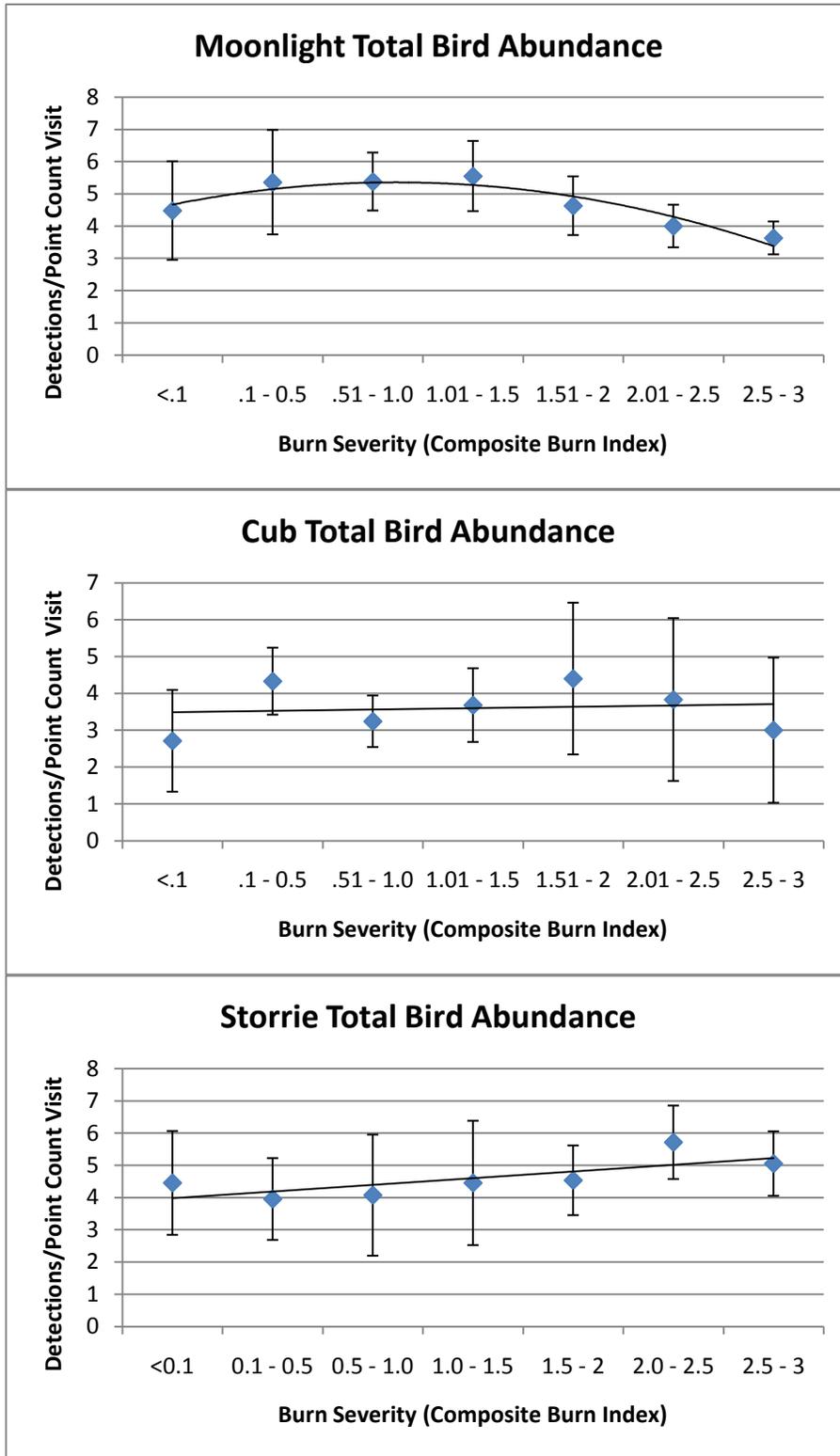


Figure 5. continued



Private Land Surveys

The private land within both the Moonlight and Storrie fires had lower indices of diversity and abundance in 2010 than Forest Service land in these fires (Figure 4). In the Moonlight Fire, species richness on private land averaged 2.4 fewer species and 2.1 fewer total birds per point than Forest Service land. The pattern was similar for the Storrie fire where private land supported 2.6 fewer species and 2.4 total birds per point compared to Forest Service land in the fire.

Using the same 30 species discussed above, we compared their abundance between private land and Forest Service land in the Moonlight and Storrie fires (Figure 6). In the Moonlight Fire, 24 of the 30 species were more abundant on Forest Service land with 12 of these differences statistically significant. Two species - Cassin's Finch and American Robin - were equally abundant on private and public land and four species; Black-backed Woodpecker, Dusky Flycatcher, Mountain Bluebird, and Oregon Junco were more abundant on private land but none of these differences were statistically significant. In the Storrie Fire, 25 species were more abundant on Forest Service land – six of which were significantly different. Three species were equally abundant on Forest Service and private land and two species - Fox Sparrow and Oregon Junco - were more abundant on private land though neither of those differences was significant.

Nine of the 50 point count stations on private land randomly fell within riparian habitat areas. Avian species richness and total bird abundance were significantly higher within these unsalvaged riparian areas across the Moonlight and Storrie fires in 2010 (Figure 7). The values observed in riparian areas were similar to those found on Forest Service land.

Figure 6. An index of the abundance (detections <50m of observers per station per visit) for each of the 30 species investigated that showed a significant difference between Forest Service and private land in the Moonlight and Storrie fires in 2010. Error bars represent one standard error.

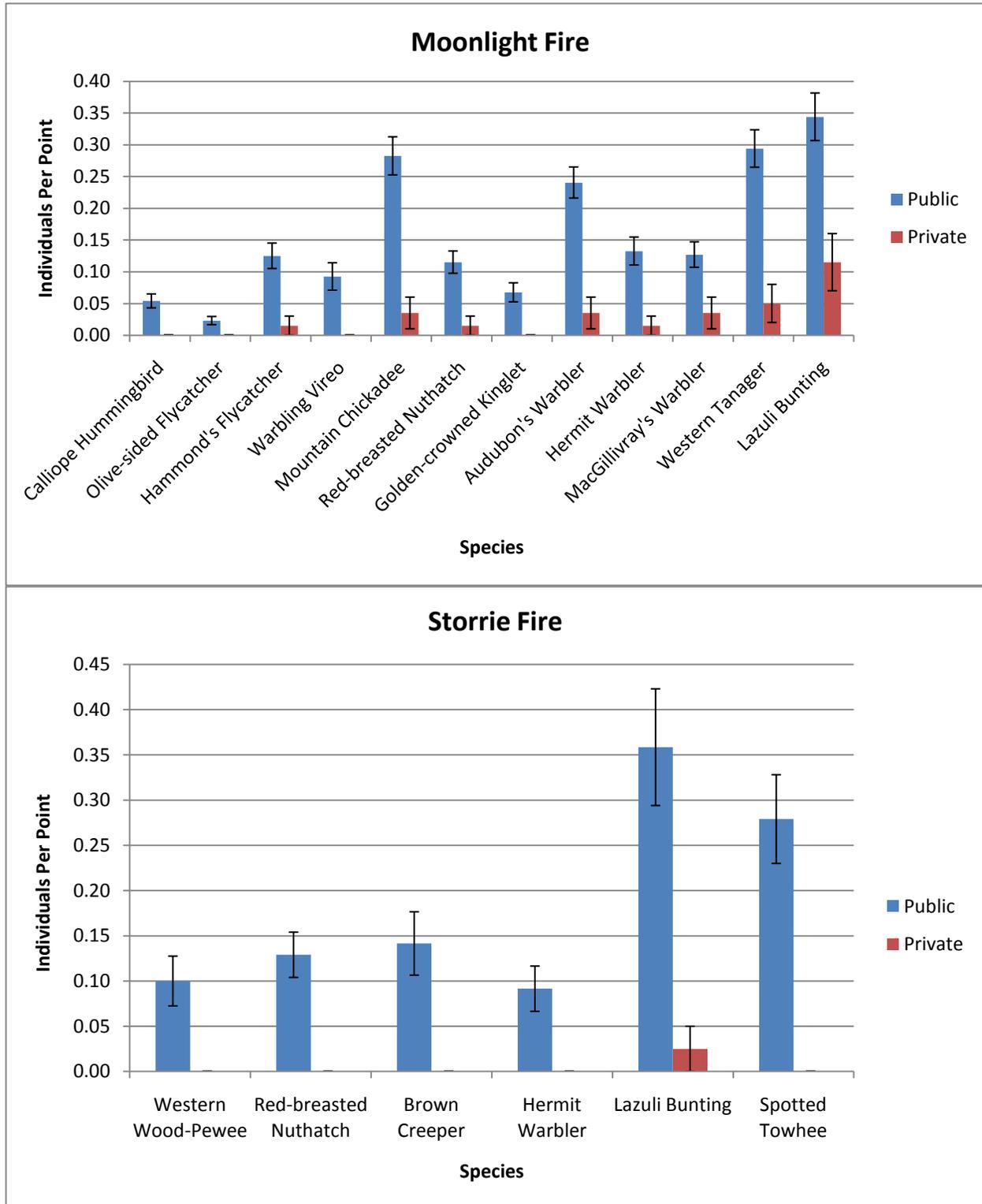
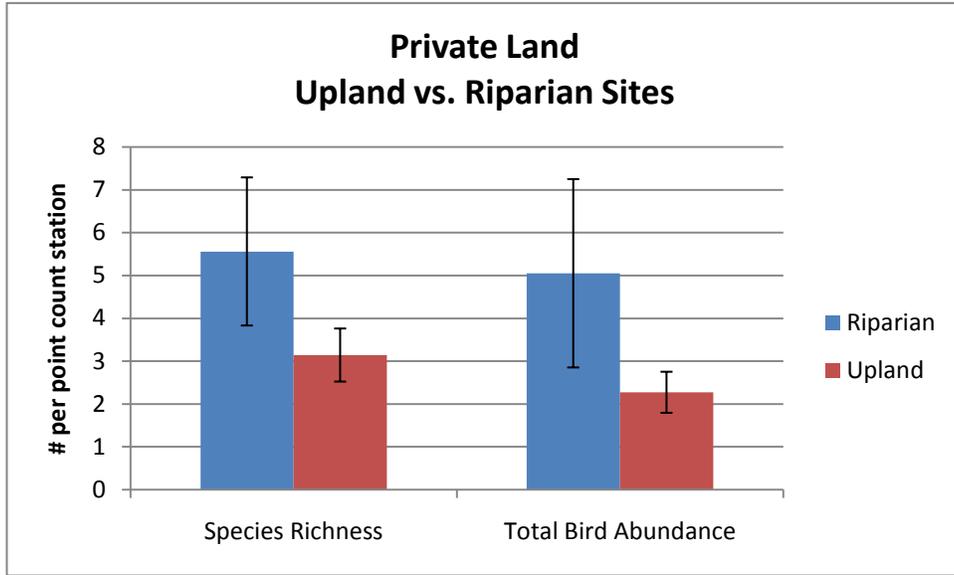


Figure 7. Species richness and total bird abundance per point at upland sites and untreated riparian areas on private land in the Storrie and Moonlight fires in 2010 with 95% confidence intervals.



Cavity Nests

A total of 180 active cavity nests of 11 species were located and confirmed active in 2010, of which 16 were outside the boundaries of the nest plots. 35 nests were found in the Cub fire, 111 in the Moonlight Fire, and 34 in the Storrie fire. In the Cub fire we found more nests for Hairy Woodpecker (10) than any other species, while we found more nests for Hairy Woodpecker and Mountain Bluebird (28 each) in the Moonlight Fire, and more for White-headed Woodpecker and Northern Flicker (9 each) in the Storrie fire (Table 2).

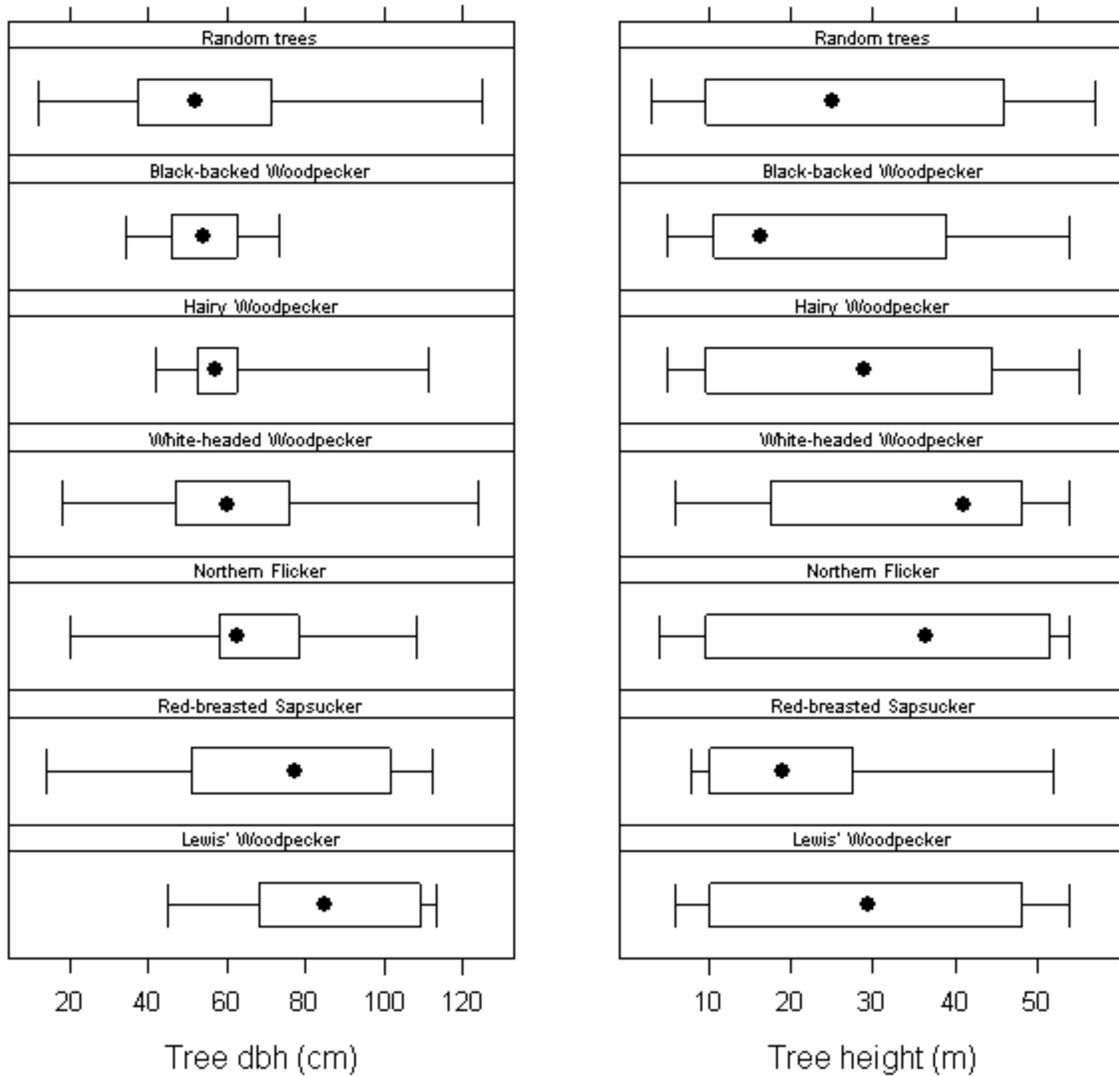
The nest trees selected were quite variable in both size and species. The diameter at breast height (DBH) ranged from 16 cm (6.6 inches) to 125 cm (49.2 inches) and the height off the ground ranged from 0.2 -- 29 m. Most woodpecker species did not show preference for nest tree size with the exception of Northern Flicker and Lewis' woodpecker which showed some preference for larger diameter trees. Due to a small sample size for Lewis' Woodpecker (n = 12) the difference was only statistically significant for Northern Flicker (Figure 8).

Table 2. Number of cavity nests confirmed by species and fire in the Plumas-Lassen Study 2009 - 2010.

	Cub		Moonlight		Storrie		Total
	2009	2010	2009	2010	2009	2010	
American Kestrel	0	0	0	1	0	0	1
Black-backed Woodpecker	1	8	3	8	0	0	20
Hairy Woodpecker	5	10	6	28	1	4	54
White-headed Woodpecker	2	8	6	14	2	9	41
Lewis' Woodpecker	0	0	1	4	2	5	12
Northern Flicker	3	5	5	14	5	9	41
Pileated Woodpecker	0	1	0	1	0	0	2
Red-breasted Sapsucker	0	0	6	9	1	3	19
Williamson's Sapsucker	0	0	1	1	0	0	2
Mountain Bluebird	0	3	9	28	1	2	43
Western Bluebird	0	0	1	3	3	2	9
Total	11	35	38	111	15	34	244

There was more evidence for preference for tree decay which is shown by both decay class and top condition of the nest tree (Figure 9). The vast majority of the nests were in dead trees (decay class 3 or greater) despite roughly a third of the “available” trees being alive. Hairy Woodpecker, Black-backed Woodpecker, and Red-breasted Sapsucker chose mostly trees of decay class 3 and 4, while White-headed Woodpecker and Lewis’ Woodpecker chose trees of higher decay classes; though White-headed Woodpecker choice was extremely variable. Northern Flicker chose mostly decay class 4 trees. Most Woodpeckers chose nest trees with broken tops with the exception of Black-backed Woodpecker which chose mostly trees with intact tops. Between 2009 and 2010, the number of Black-backed Woodpecker nests found increased from 4 to 16 and the percentage of those nests that were found in intact trees increased from zero to 14.

Figure 8. Median (dot), 25-75% interquartile range (box), and extremes (bars) in nest tree diameter (DBH) and nest tree height for six woodpecker species and randomly selected trees in the Storrie, Moonlight, and Cub fires in 2009 and 2010.



Nests were found in eight tree species/species groups in 2009 and 2010. However, the majority of nests were found in five species/species groups: true fir (both red and white), yellow pine (includes both Ponderosa and Jeffrey), and Douglas-fir. These were some of the most common tree species available based on our sample of random trees. However, incense cedar was also one of the more common available trees, though only one nest (White-headed Woodpecker) was found in a cedar. Black-backed Woodpecker almost exclusively selected true fir and pine species, while Hairy Woodpecker, White-headed Woodpecker, and Northern Flicker

were less particular with the relative frequencies of use reflecting what was available. Over a quarter of the Red-breasted Sapsucker nest were found in quaking aspen despite this tree species having an extremely small proportion of the available trees. Similarly, Lewis' Woodpecker chose Douglas-fir as a nest tree more than it was available, while none of the nine Lewis' Woodpecker nests were in true Fir, the most prevalent tree species (Figure 10).

Figure 9. The decay class and top condition of the nests of 6 woodpecker species compared to the sample of randomly selected trees across three fires in the Northern Sierra Nevada in 2009 and 2010. BA=Broken After Fire, BB=Broken Before Fire, DT=Dead Top, F= forked, I= Intact.

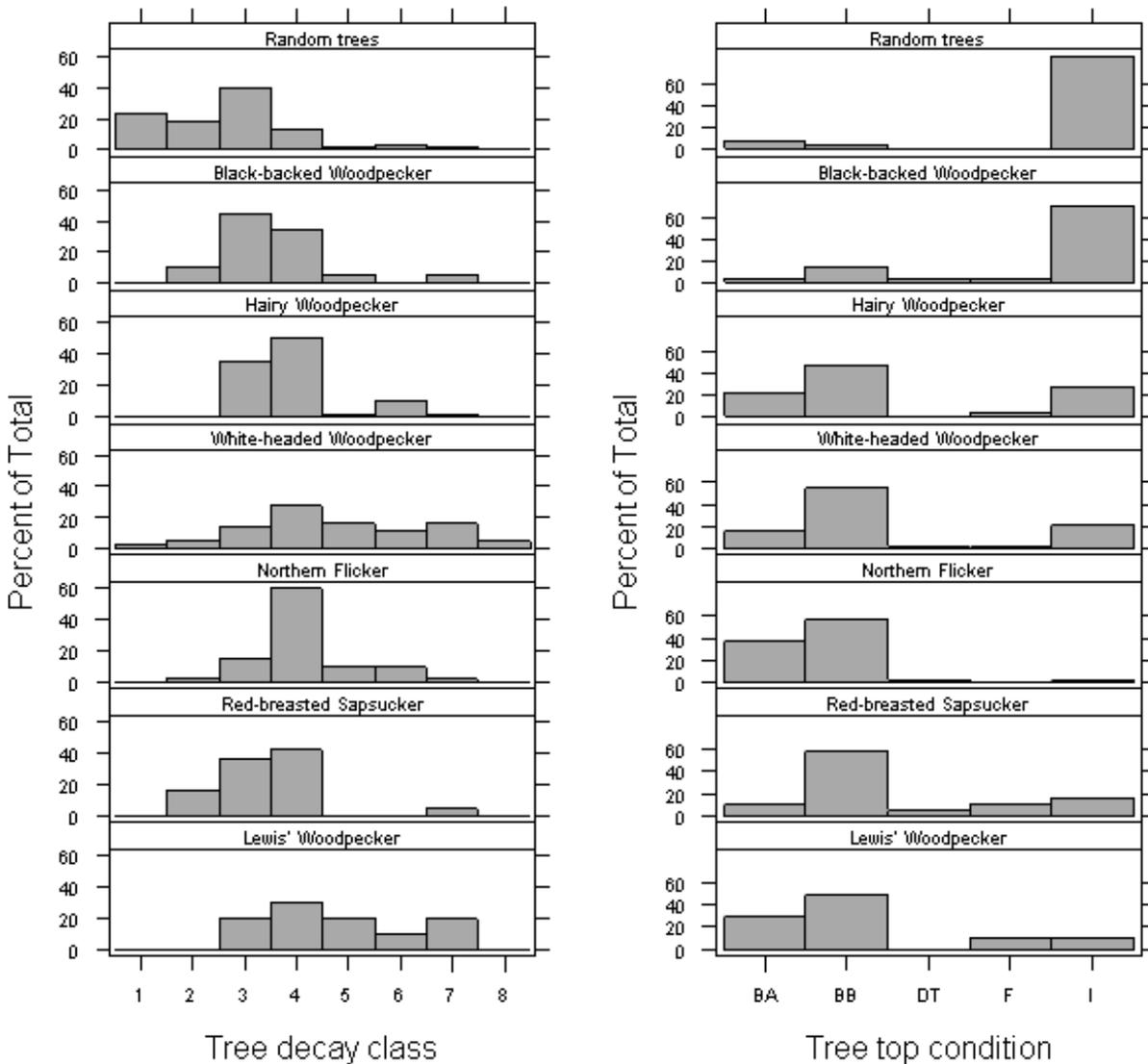
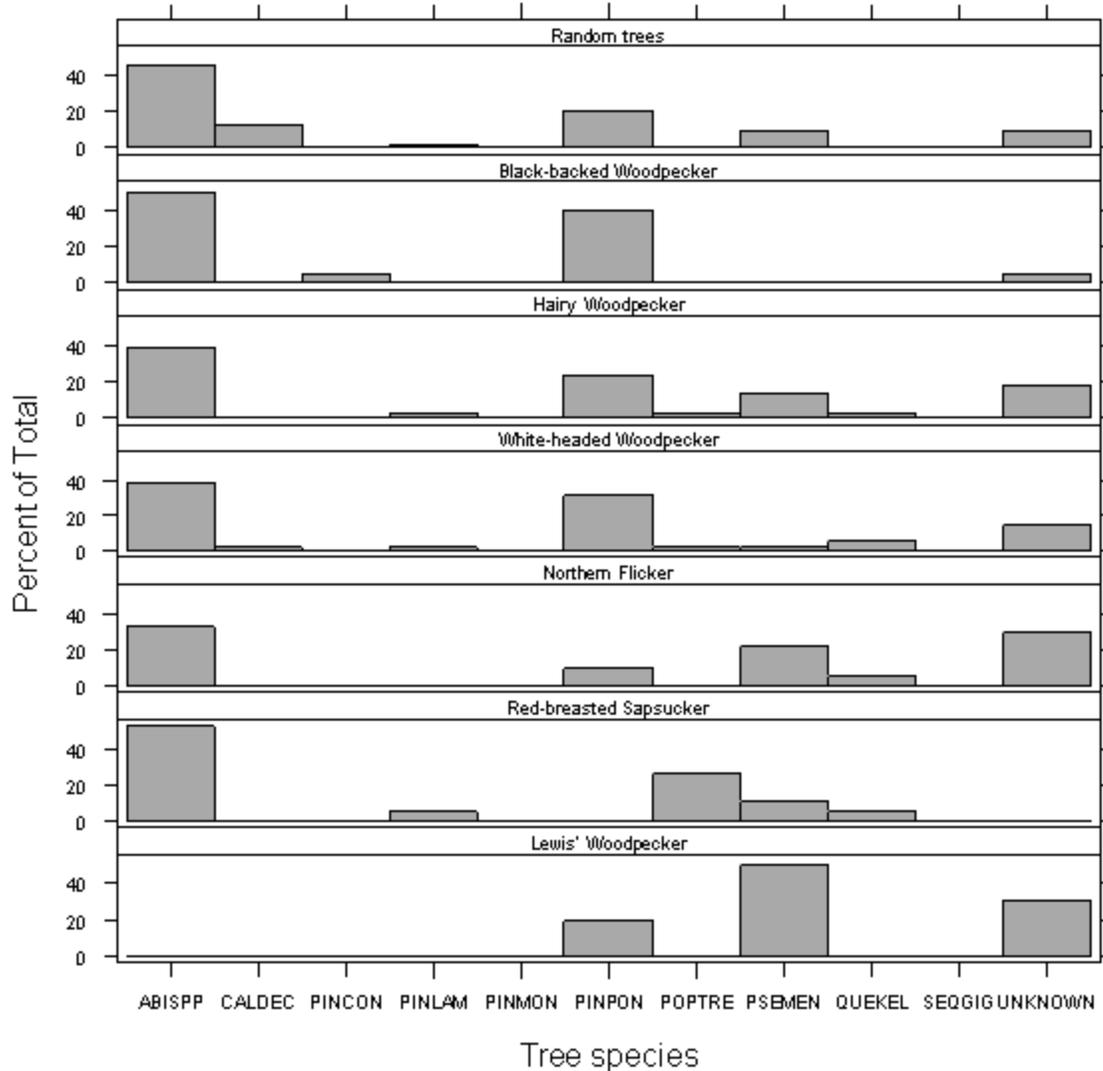


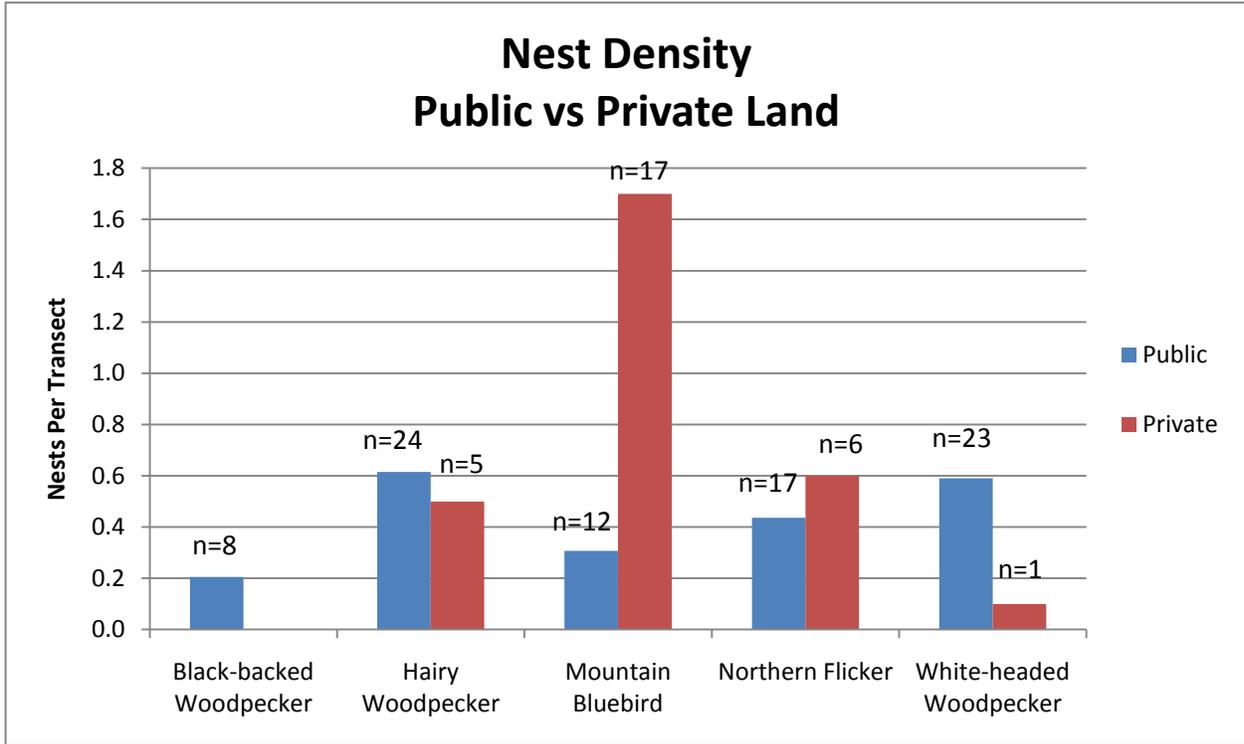
Figure 10. Nest tree species for 6 woodpecker species compared to the sample of random trees across three fires in the Northern Sierra Nevada in 2009 and 2010. ABISPP = true fir, CALDEC = incense cedar, PINCON = lodgepole pine, PINLAM = sugar pine, PINMON = western white pine, PINPON = ponderosa pine, POPTRE = quaking aspen, PSEMEN = Douglas-fir, QUEKEL = black oak, SEQGIG = giant sequoia.



We compared differences in densities of cavity nests found between public and private land in the Moonlight and Storrie Fires (Figure 11). Sixteen Black-backed Woodpecker nests were found on Forest Service land whereas none were found on private land. Hairy Woodpecker and White-headed Woodpecker were also found nesting in higher densities on Forest Service land. However, slightly more Northern Flicker nests per transect were found on private land. The nest density of Mountain Bluebird accurately reflects the point count data,

showing a much higher abundance on private land. Furthermore, 15 of the 17 Mountain Bluebird nests found on private land were in the Moonlight Fire.

Figure 11. Comparison of the number of nests confirmed per transect on public versus private land in the Moonlight and Storrie Fires in 2010.



Discussion

Avian Community Composition Burned vs. Green Forest

Avian community composition between fires and green forest in the study area varied substantially when we considered the abundance of 30 species, but 29 of the 30 species did occur in both green forest and at least one of the fires. The exception was Black-backed Woodpecker which was not recorded in the green forest in 2010. However, this species does occur in green forest in the PLAS study area outside of areas that have recently burned, but at much lower densities than within recently burned forest in the Sierra Nevada (PRBO data, Siegel et al. 2010).

We found higher per point indices of abundance and diversity in green forest than any of the three fire areas but as a whole the fire areas supported equal or greater total bird abundance per unit area than the surrounding green forest in 2009 and 2010. Green forest in our study area likely have greater structural diversity and total foliage volume at the point level (50 m radius), which can support a greater diversity of species (McCarthy et al. 1966, Verner and Larson 1989), while post-fire habitats in our study area appear more heterogeneous on a landscape scale. Therefore, whether green forest or post-fire habitat supports greater avian diversity depends on the spatial scale at which the question is considered.

Our data shows a clear signal of woodpeckers, as well as shrub, ground, and edge-associated species, being more abundant in fire areas, and foliage-gleaning and open-cup tree-nesting species being more abundant in green forest habitat; consistent with previous studies in the Sierra Nevada (Raphael et al. 1987). These patterns appear to get stronger as the time since fire and burn severity increase such that the most recent and lowest severity Cub fire is most similar to the green forest community composition and the oldest and high severity Storrie fire is most different, especially for shrub associated species.

An understanding of the differences in avian community composition and specifically the relative abundance of species between unburned forest, mechanically treated habitat, and post-fire habitats can help guide the management of these areas. With high severity areas of post-fire habitat having lower densities of late seral associated species these areas might best be prioritized for sustaining populations of early successional species. Likewise, later seral habitat areas are probably not the ideal location for creating large quantities of Fox Sparrow habitat. However, this does oversimplify things as habitat mosaics are also an important part of ensuring biological diversity in these forests. A better understanding of the landscape patterns and appropriate scales to manage for habitat mosaics is needed.

Time Since Fire & Burn Severity

The effects of burn severity on the avian community varied across the three fires. Other factors such as time since fire, landscape patterns of severity, and pre-fire habitat conditions are likely interacting with severity to influence the avian community at the 50 m scale at which

we sampled it, as each of the three fire areas showed different patterns. Thus, it is hard to determine if the increasing linear effect of severity found in the Storrie fire or the curvilinear effect in the Moonlight fire are a result of time since fire or something more complex such as landscape patterns of burn severity and pre-existing habitat conditions. Monitoring the two younger fires for several more years would help answer these questions. Additionally, future analysis of the effects of severity with co-variables of time since fire, patch size and pattern, and pre-fire habitat conditions will give us a better understanding of the effects of fire severity on the avian community and hence more information to help guide both pre-and post-fire habitat restoration activities.

Despite the limitations of this analysis, there are some interesting findings that merit discussion. There is some evidence from the Moonlight and Cub -and less so from the much older Storrie fire - that moderate severity areas support the greatest abundance and diversity of avian species. Across each of the three fires, the lowest species richness was recorded for the highest severity class but, for the Cub and Storrie Fires the unburned areas within the fire perimeter also had relatively low species richness. These results suggest that managing for mixed severity wildfire with the majority of the landscape burning at moderate severity may maximize local and landscape level avian diversity. Further investigation of the landscape scale patterns of fire severity and patch size will help better answer this important question.

The effects of fire severity and time since fire are most apparent when comparing the abundance of individual species; with those favoring fire killed trees, shrubs, and bare ground most abundant in high severity areas and foliage gleaning tree nesting species associated with low severity areas. A number of species were far more abundant in the Storrie Fire than the two younger fires and in both the Storrie and Moonlight fire compared to the lower severity Cub fire. Species such as Nashville Warbler which is strongly associated with Black Oak in the region (Burnett and Humple 2003), are readily using the substantial oak regeneration found throughout the fire. Two declining species, Olive-sided Flycatcher and Western Wood-Pewee, forage on flying insects at habitat edges (Bemis & Rising 1999, Altman and Sallabanks 2000). They were both more abundant in the Storrie Fire than either of the other fires or green forest. The hard edges created by the Storrie fire along with the substantial patches of green forest (at

least in the area of the fire we were able to survey) are likely the reasons for their elevated abundance here. It is not clear what the effect of time since fire is on these two species as there is debate whether the association these species have with fire areas is due to the physical habitat conditions it creates or the increase in prey base it brings about (Meehan and George 2003, Roberston and Hutto 2007).

In 2009, Fox Sparrows were less abundant in the Moonlight Fire than green forest, but rapid regeneration of the shrub layer between 2009 and 2010 in the Moonlight fire seems to have promoted a higher abundance of Fox Sparrows than green forest, though time-dependent trends such as this are difficult to discern with just two years of data. We observed similar patterns for several other shrub-dependent species including MacGillivray's Warbler and Spotted Towhee, which are 1.4 and 4 times more abundant respectively in the Storrie Fire than green forest. These large patches of dense shrub cover that will develop in high severity patches of the Moonlight and Cub fires will support a number of species and, when interspersed with areas of green forest, are a key component for maximizing avian diversity in post-fire environments.

Information about colonization rates and how long after a fire the abundance of these species persists at maximum levels can be used to determine appropriate re-entry rotations for managing habitat following fire. Based on our results and observations made of habitat conditions within the fires, there is a approximately a five year lag before dense shrub habitats form that maximize densities of species such as Fox Sparrow and MacGillivray's Warbler. Nine and ten years after burning the Storrie fire supported nearly double the abundance of Fox Sparrow – the indicator for montane chaparral habitat in the Sierra Nevada – than the Moonlight fire did 2 to 3 years after burning. Our results suggest that 10 years after fire the early successional shrub habitats in the Storrie fire are supporting very high densities of shrub dependent birds and are showing no sign of senescing. A re-entry rotation of 20 - 30 years for chaparral habitat may maximize abundance of species such as Fox Sparrow and would mimic the historic fire return interval for this habitat in the Sierra Nevada (Barbour and Major 1988).

Results from the PLAS green forest study suggests the use of prescribed fire has far more positive effects on the avian community compared to the use of mechanical mastication

in shrub habitats in the region (Burnett et al. 2009). If mechanical mastication is used, especially in high quality shrub bird habitat - as currently exists in the Storrie fire – retaining leave islands of very dense shrubs will help provide nesting habitat and reduce negative impacts to shrub dependent species. However, best management practices for these species would be to avoid disturbing this habitat for a number of years beyond the ten since the Storrie fire burned.

Post-fire Cavity Nest Characteristics

In 2010 we found three times as many nests as in 2009 with 20% greater sampling effort. Since there were similar increases across each of the three fires we are hesitant to assume this represents a real increase in densities of these species on our plots. We had observers with more experience finding nests in 2010 with the return of two of our crew members from 2009. However, previous studies in western U.S. forests have found cavity nesting species densities were greatest four to five years following fire (Saab and Dudley 1998, Saab et al. 2004). It will be interesting to compare these nest numbers to 2011 as the Moonlight fire reaches four years post burn and the Cub three.

The importance of post-fire habitats for cavity-nesting and bark-foraging birds is well established (Raphael et al. 1987, Hutto 1995, Saab and Dudley 1998). However, little information exists for the Sierra Nevada describing the important characteristics in post-fire snag-dominated habitats that determine the density and diversity of cavity nesting species. Our results here provide some of the only detailed information for a whole suite of cavity nesting species in post-fire habitats in the Sierra Nevada (also see Raphael and White 1984).

Patterns in nest tree characteristics found in 2010 affirm those found in 2009 with a wide variety of tree species being used across bird species. Higher decay classes and broken top snags are still being used more than they are available. Though, in 2010 we started to see more birds using the fire killed snags for nesting than they did in 2009 suggesting those snags are now becoming suitable for nesting. Black-backed woodpecker continued to show very little selection for tree species, size, or decay class in 2010. They readily used what was available on the landscape suggesting other factors are driving this patchily distributed, relatively uncommon species occupancy of habitat within these fires. Other studies have suggested that

temporally limited food resources are the most important factor for determining this species distribution in high severity fire areas (Saab and Dudley 1998, Dixon and Saab 2000).

Private vs. Public Land

The differences in post-fire management approaches on public and private land within the Moonlight and Storrie fires is evident from space (Google earth). As of 2010, the majority of public land had been untouched since the fire, whereas private land has been extensively salvaged, prepped, and replanted. As such, the differences in the avian community between the two were considerable.

Salvage logging of areas affected by moderate to high severity fire, as has occurred on private land in both fires, can substantially reduce the number of snags available for a number of woodpecker species for both nesting and foraging (Hutto 2006, Saab 2009). Differences in nest densities of Black-backed Woodpecker, White-headed Woodpecker, and to a lesser extent Hairy Woodpecker, between public and private land, suggest that management approaches on private land are resulting in less suitable habitat for a number of cavity nesting species. Interestingly, Mountain Bluebird appeared to strongly prefer private land in the Moonlight fire. This species forages primarily on the ground and is found in open country, both features far more common on private land. Additionally, Lewis' Woodpecker nests were found almost exclusively on the boundaries between public and private land or in un-salvaged riparian areas on private land. This species has been shown to respond positively to moderate levels of salvage (Saab et al. 2007).

Analysis of point count data also suggests that private lands support a significantly less diverse and abundant avian community than the surrounding Forest Service land that has largely been left untouched since the fire (at least through spring 2010). Interestingly, a number of foliage-gleaning green tree associated species were those found to be significantly less abundant on private land. This is likely a result of severity being higher on private land but also a result of private land management practices that err on the side of taking trees that might survive versus leaving trees that might die. This results in little if any overstory trees left and

thus little habitat for the diverse group of tree-nesting foliage-gleaning species such as Hermit Warbler and Red-breasted Nuthatch.

The other guild of species significantly less abundant on private land was the shrub associates. It is not as surprising a result to find species associated with dense patches of shrubs less abundant on private land. High severity habitat on public land is typically characterized by a dense shrub layer that regenerates quickly after the disturbance. However, on private land measures have been taken to reduce shrub regeneration (J. Pudlicki pers. comm.). Shrub dependent species such as Calliope Hummingbird, Lazuli Bunting, MacGillivray's Warbler, and Spotted Towhee were exceedingly rare on the private land sites in 2010. Lazuli Bunting showed the largest differences and the effect was consistent across the two fires. A lack of suitable dense shrub nesting substrate along with a lack of herbaceous vegetation - likely a result of the use of herbicides - that provide seeds and support insect prey for this species are likely the reasons Lazuli Bunting are scarce on private land in these fires. Additionally, the dense stands of planted conifers now over 2m tall in the Storrie fire is probably not to the liking of this open habitat associated species.

Olive-sided Flycatcher and Western Wood-pewee, two species shown to be declining in the Sierra Nevada, were not detected on private lands despite being more abundant in post-fire areas. These birds are not typically associated with one particular habitat, but rather the juxtaposition of unlike habitats. Management practices that produce homogenous landscapes such as those on private land in the Storrie and Moonlight fires are unlikely to support these and other species associated with habitat mosaics. Additionally, as these even-aged stands develop over time they are unlikely to provide habitat for a wide range of species associated with stand and landscape level heterogeneity.

The riparian areas within private land were not entered during salvage operations or subsequent herbicide treatments in both fires resulting in far more snags, live trees, shrubs, and herbaceous vegetation. These areas supported the greatest number of species and densities of cavity nesting birds on private land in the two fires. Leaving more patches of un-salvaged habitat on private land would undoubtedly increase avian diversity in the short term and help promote a more heterogeneous forest in the future.

Conclusions

In the Sierra Nevada, considerable debate surrounds the management of post-fire habitat. After nearly a century of fire suppression policies in the Sierra Nevada, the area affected by wildfire each year appears to be increasing back towards pre-suppression levels (Miller et al. 2008). Thus, there is a growing need to understand the value of the habitats created by wildfire and the critical elements required by the unique and relatively diverse avian community in the Sierra Nevada.

Wildfires provide a unique opportunity to mold a landscape into the forest composition that will exist there for the decades to come. However, post-fire areas are not blank slates or catastrophic wastelands; they are a unique component of the ecosystem that supports a diverse and abundant avian community that should be considered in planning post-fire management. The results from this ongoing study, especially with several more years of monitoring, can provide some important information on factors influencing the unique bird assemblages in post-fire habitat in order to ensure their needs are met while meeting other post-fire objectives.

In future years we will use available remotely sensed data on burn severity and possibly LiDAR data to better understand the importance of severity class, patch size, and snag densities for the various species associated with post-fire habitat. Finally, we will compare bird assemblages within fuel reduction treatments in green forest, post-fire treatments, and untreated post-fire habitat to determine the effects of various management practices on a broad range of avian species to provide insight for a balanced approach to management of these forest ecosystems.

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Appendix A. Outreach and Publications

Published

Nesting Ecology of Yellow Warblers in Montane Chaparral Habitat in the Northern Sierra Nevada – Western North American Naturalist 70(3):355-363

In prep

Short-term response of avian species to fuel treatments in the Sierra Nevada – draft submitting for publication spring 2011.

Black-backed Woodpecker nest tree selection in the Sierra Nevada, California – draft submitting for publication February 2011

Woodpecker nest tree selection in Sierra Nevada post-fire environments: Is there evidence for spatial or temporal heterogeneity? – draft submitting for publication March 2011.

Presentations

Managing Post-fire Habitat for Birds in the Sierra Nevada – oral presentation at the Annual PLAS symposium Quincy, CA April 2010

Managing post-fire habitat for birds in the Sierra Nevada – poster presentation – Pre and Post-Wildfire Forest Management for Ecological Restoration and Fire Resiliency Conference –Sacramento, CA 2/10/10.

PRBO's Sierra Nevada Program – Management Indicator Species Online Tools – Regional Biologist Training – Susanville, CA. 2/2/10

Other Outreach

“Birds in the Park” – presentation on managing coniferous forest for birds and bird banding demonstration in collaboration with Lassen Volcanic National Park – over 200 park visitors participated 7/18/10.

“Habitat for Birds and Humanity” – Sierra Institute for Community and Environment sponsored field trip. – 6/26/2010

Participated in several Lassen National Forest field trips to discuss fuel reduction projects

Attended the Lassen National Forest Storrie Fire Science symposium – Susanville, CA March 2010

Appendix B. PRBO's Plumas-Lassen Study sites with dates surveyed in 2010.

Transect Name	Transect Short Name	Visit 1	Visit 2
213	213	6/2/2010	6/17/2010
214	214	6/5/2010	6/22/2010
222	222	6/6/2010	6/24/2010
224	224	5/18/2010	6/8/2010
313	313	6/3/2010	6/18/2010
322	322	6/3/2010	6/18/2010
413	413	6/20/2010	6/28/2010
414	414	5/19/2010	6/15/2010
422	422	6/2/2010	6/18/2010
423	423	5/16/2010	5/16/2010
424	424	5/19/2010	6/9/2010
Blackhawk Creek 1	BLH1	5/16/2010	6/15/2010
Butt Valley Reservoir 3	BVR3	6/3/2010	6/24/2010
Caribou 2	CAR2	6/6/2010	6/23/2010
Caribou Transect 3	CAR3	6/6/2010	6/23/2010
Cub 1	CB01	5/28/2010	6/14/2010
Cub 10	CB10	6/6/2010	7/1/2010
Cub 11	CB11	6/23/2010	7/1/2010
Cub 12	CB12	5/21/2010	N/A
Cub 13	CB13	5/21/2010	6/14/2010
Cub 14	CB14	5/21/2010	6/14/2010
Cub 2	CB02	5/25/2010	6/14/2010
Cub 3	CB03	6/8/2010	6/17/2010
Cub 4	CB04	5/25/2010	6/22/2010
Cub 5	CB05	6/22/2010	7/1/2010
Cub 7	CB07	6/29/2010	7/1/2010
Cub 8	CB08	6/29/2010	6/30/2010
Cub 9	CB09	6/6/2010	6/23/2010
DFPZ - TU4	D401	6/14/2010	6/25/2010
DFPZ - TU4	D402	5/18/2010	6/7/2010
DFPZ - TU4	D403	5/19/2010	6/7/2010
DFPZ - TU4	D404	5/19/2010	6/9/2010
DFPZ - TU4	D405	5/15/2010	6/8/2010
DFPZ - TU4	D407	6/1/2010	6/26/2010
DFPZ - TU4	D408	6/19/2010	6/28/2010
Halstead Valley3	HAL3	5/27/2010	6/13/2010
Lower Knox Flat Transect 1	LKF1	5/19/2010	6/13/2010
Lower Knox Flat Transect 3	LKF3	6/2/2010	6/18/2010

Meadow Valley 1	MVY1	5/18/2010	6/8/2010
Meadow Valley 2	MVY2	6/18/2010	6/29/2010
Meadow Valley Owl	MVYW	6/18/2010	6/29/2010
Miller Fork 1	MIF1	6/14/2010	6/25/2010
Miller Fork 2	MIF2	6/19/2010	6/28/2010
Miller Fork 3	MIF3	6/20/2010	6/28/2010
Moonlight 1	ML01	5/20/2010	6/20/2010
Moonlight 10	ML10	6/20/2010	6/27/2010
Moonlight 11	ML11	5/21/2010	6/9/2010
Moonlight 12	ML12	6/6/2010	6/24/2010
Moonlight 13	ML13	5/21/2010	6/9/2010
Moonlight 14	ML14	6/19/2010	6/26/2010
Moonlight 15	ML15	5/24/2010	6/10/2010
Moonlight 16	ML16	5/29/2010	6/13/2010
Moonlight 17	ML17	6/20/2010	6/29/2010
Moonlight 18	ML18	5/31/2010	6/25/2010
Moonlight 19	ML19	6/19/2010	6/25/2010
Moonlight 2	ML02	5/29/2010	6/26/2010
Moonlight 20	ML20	5/30/2010	5/30/2010
Moonlight 21	ML21	5/31/2010	6/24/2010
Moonlight 22	ML22	5/31/2010	6/24/2010
Moonlight 23	ML23	5/29/2010	6/26/2010
Moonlight 24	ML24	5/24/2010	6/10/2010
Moonlight 26	ML26	5/29/2010	6/13/2010
Moonlight 27	ML27	5/20/2010	6/10/2010
Moonlight 3	ML03	5/31/2010	6/25/2010
Moonlight 4	ML04	5/21/2010	6/13/2010
Moonlight 5	ML05	5/20/2010	6/10/2010
Moonlight 6	ML06	5/20/2010	6/13/2010
Moonlight 7	ML07	6/6/2010	6/24/2010
Moonlight 25	ML25	6/5/2010	6/20/2010
Moonlight 28	ML28	5/30/2010	6/16/2010
Moonlight 29	ML29	5/30/2010	6/16/2010
Moonlight 30	ML30	5/30/2010	6/16/2010
Moonlight 31	ML31	5/30/2010	6/16/2010
Moonlight 32	ML32	6/5/2010	6/21/2010
Moonlight 33	ML33	6/5/2010	6/21/2010
Moonlight 9	ML09	5/30/2010	6/26/2010
Ohio Creek 2	OHC2	6/2/2010	6/17/2010
Pine Leaf Creek Transect 1	PLC1	5/18/2010	6/7/2010
Seneca 1	SEN1	5/18/2010	6/8/2010
Silver Lake 1	SIL1	6/22/2010	6/30/2010
Silver Lake 2	SIL2	6/22/2010	6/30/2010

Silver Lake 3	SIL3	5/16/2010	6/8/2010
Silver Lake Owl Transect	SILW	6/1/2010	6/26/2010
Snake Lake 2	SNK2	5/15/2010	6/7/2010
Snake Lake 3	SNK3	5/15/2010	6/7/2010
Snake Lake Owl Transect	SNKW	5/19/2010	6/7/2010
Soda Creek Transect 1	SOD1	6/2/2010	6/15/2010
Soda Creek Transect 2	SOD2	6/2/2010	6/15/2010
Spanish Creek 2	SPC2	6/1/2010	6/15/2010
Storrie 1	ST01	5/25/2010	6/16/2010
Storrie 10	ST10	5/24/2010	6/11/2010
Storrie 11	ST11	5/25/2010	6/11/2010
Storrie 12	ST12	5/31/2010	N/A
Storrie 2	ST02	5/24/2010	6/10/2010
Storrie 3	ST03	5/24/2010	6/10/2010
Storrie 4	ST04	N/A	N/A
Storrie 5	ST05	6/23/2010	7/1/2010
Storrie 6	ST06	6/29/2010	7/1/2010
Storrie 7	ST07	6/23/2010	6/30/2010
Storrie 9	ST09	5/24/2010	6/22/2010
Storrie 13	ST13	5/25/2010	6/11/2010
Storrie 15	ST15	N/A	N/A
Storrie 16	ST16	6/1/2010	6/17/2010
Storrie 17	ST17	6/1/2010	6/17/2010
Storrie 18	ST18	6/1/2010	6/16/2010
Storrie 19	ST19	6/1/2010	6/18/2010

Appendix C. Visual representation of qualitative scale used for categorizing tree/snag decay (from Hunter 1990).

