



Influences of postfire salvage logging on forest birds in the Eastern Cascades, Oregon, USA

Rebecca E. Cahall^{*}, John P. Hayes¹

Department of Forest Science, Oregon State University, Corvallis, OR 97331, United States

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ABSTRACT

In coniferous forests of western North America, fire is an important disturbance that influences the structure and composition of floral and faunal communities. The impacts of postfire management, including salvage logging and replanting, on these forests are not well known. We compared densities and relative abundances of forest birds after fire in unsalvaged stands and stands subjected to one of two intensities of salvage logging (moderate, 30 snags retained per ha and heavy, 5–6 snags retained per ha) in mixed-conifer forests in central Oregon. We used analysis of variance with repeated measures to evaluate three hypotheses concerning the influence of different intensities of salvage on densities or relative abundances of sixteen species of birds, and two hypotheses concerning the influence of time since salvage logging on relative abundances or densities of birds. We also examined the relationship between vegetation and abundances of each bird species. We did not detect significant differences among treatments in densities or relative abundances for eight species and one genus of birds. We detected significant differences for seven species, though the patterns differed among species. Relative abundances or densities of the black-backed woodpecker (*Picoides arcticus*), hairy woodpecker (*P. villosus*), brown creeper (*Certhia americana*), western wood-pewee (*Contopus sordidulus*) and yellow-rumped warbler (*Dendroica coronata*) were lower in the heavy and moderate salvage treatment compared to the unsalvaged treatment, while densities of the dark-eyed junco (*Junco hyemalis*) and fox sparrow (*Passerella iliaca*) were greater in the moderately and heavily salvaged stands than in the unsalvaged treatment. We detected significant differences between years for four species of birds. Our findings suggest that both cavity-nesting and cup-nesting species respond to salvage logging, and that some species respond uniquely to habitat features influenced by salvage logging. For species that responded negatively to salvage logging, the moderate salvage intensity did not appear to mitigate the negative influence of salvage logging. Areas of unlogged burned forest appear to provide important habitat for some species of birds following forest fires. Our findings parallel those of other recent studies of these species, suggesting robust patterns that transcend particular locations.

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1. Introduction

Avian communities in postfire forests are unique (Rocky mountains: Hutto, 1995; Caton, 1996; Kotliar et al., 2002; Sierra Nevada: Bock and Lynch, 1970; boreal: Hobson and Schieck, 1999; Morissette et al., 2002; Schieck and Song, 2006). Cavity-nesting birds (Hutto, 1995; Saab and Dudley, 1998; Kotliar et al., 2002), aerial insectivores (Hutto, 1995; Caton, 1996; Kotliar et al., 2002), and ground- and shrub-foraging birds (Bock and Lynch, 1970; Caton,

1996) often increase in abundance following fire, and burned forests may be necessary for the persistence of some species of birds (Raphael and White, 1984; Raphael et al., 1987; Hutto, 1995; Murphy and Lehnhausen, 1998; Hobson and Schieck, 1999; Hoyt and Hannon, 2002). The abundance and characteristics of snags (Everett et al., 2000; Smith, 2000), abundance of insect prey (Muona and Rutanen, 1994; Rasmussen et al., 1996; McHugh et al., 2003), and characteristics of the forest floor and herbaceous and shrub communities (Kauffman, 1990; Agee, 1993; Smith, 2000) following stand-replacing fire are important habitat elements contributing to responses of wildlife to postfire conditions.

Although postfire forests provide habitat for many species, there often are competing management objectives in burned forests. Salvage logging, the removal of dead or dying trees from a landscape after a disturbance (Gorte, 1996), is often proposed or implemented to meet economic goals. Salvage logging directly or

^{*} Corresponding author. Tel.: +1 541 760 9849.

E-mail addresses: rebecca.cahall@earthlink.net (R.E. Cahall), hayesj@ufl.edu (J.P. Hayes).

¹ Present address: Department of Wildlife Ecology and Conservation, University of Florida, Gainesville, FL 32611, United States. Tel.: +1 352 846 0552; fax: +1 352 392 6984.

indirectly influences almost every aspect of postfire habitat that shape wildlife response (Karr et al., 2004; Donato et al., 2006; Lindenmayer and Ough, 2006; Russell et al., 2006), and as a result can profoundly influence presence and abundance of many species of wildlife.

A number of studies have examined the response of populations of birds to intensive postfire salvage that removes all or nearly all of the snags from an area (e.g., Saab and Dudley, 1998; Hutto and Gallo, 2006; Saab et al., 2007). Most of these studies have focused on cavity-nesting birds, although a few have examined the responses of the entire bird community to salvage logging (LeCoure et al., 2000; Morissette et al., 2002; Schwab et al., 2006). The influences of less intensive salvage activities on birds have received less study, and the responses of species to salvage intensity revealed in these studies has been variable. For example, the mountain bluebird (*Sialia currucoides*) and some woodpeckers (*Picoides* spp.) have greater densities or abundances in unsalvaged forests than in partially and completely salvaged forests in some studies (Saab and Dudley, 1998; Koivula and Schmiegelow, 2007; Saab et al., 2007), but greater numbers in partially salvaged forests than in unsalvaged or salvaged forests in another (Haggard and Gaines, 2001). Only two studies to date have evaluated the effects of different intensities of postfire salvage logging on cup-nesting birds (LeCoure et al., 2000; Schwab et al., 2006), both of which were conducted in boreal forests, and only one (Schwab et al., 2006) had a replicated study design.

Our objective was to determine differences in relative abundances or densities of birds in stands subjected to different intensities of salvage logging. We compared densities and relative abundance of birds among three treatments: unsalvaged, moderately salvaged (30 snags ≥ 35.6 cm DBH ha^{-1}), and heavily salvaged (5–6 snags ≥ 50.8 cm DBH ha^{-1}). We also examined the correlation of three vegetation variables with densities or relative abundances of each bird species in an attempt to better understand potential causal mechanisms responsible for differences in the response of some species.

2. Methods

2.1. Study area

We selected stands for study within the perimeter of an 8511 hectare (ha) fire that occurred on Davis Mountain, located just east of the crest of the Cascade Mountains in Deschutes County in Central Oregon. The pre-fire forest was composed primarily of ponderosa pine (*Pinus ponderosa*), with components of Douglas-fir (*Pseudotsuga menziesii*) and sugar pine (*P. lambertiana*). The stands also included lodgepole pine (*P. contorta*), white fir (*Abies concolor*), and Shasta red fir (*A. magnifica* var. *shastensis*), which were not typical species of historic forest conditions (USDA, 2004). Pre-fire land-use designations were matrix forest and late-successional reserves (LSR), as defined by the Northwest Forest Plan (USDA and USDI, 1994). The primary species of pre and postfire understory shrubs included snowbrush (*Ceanothus velutinus*), green-leaf manzanita (*Arctostaphylos patula*), and western chinquapin (*Castanopsis chrysophylla*). Pre-fire soils were comprised of a deep mantle of ash and pumice over an older layer of similar soil, and were characterized as highly permeable and well drained (USDA, 2004). Elevation ranged from 1350 to 2025 m and slopes were $\leq 25^\circ$.

The Davis Mountain fire occurred from late June through early July 2003. Within the fire perimeter, at least 95% tree mortality occurred within 75% of the burned forest. Prior to the beginning of our study, 2514 ha of the burned forest were salvage-logged between the fall of 2004 and early summer 2005. Three management prescriptions were implemented: unsalvaged, moderately salvaged, and heavily salvaged. Unsalvaged stands were located in LSR and no trees were removed from these sites. Within salvaged

stands, all snags with a diameter at breast height (DBH) >91 cm were retained. Moderately salvaged stands were located in LSR and 30 snags ≥ 35.6 cm DBH ha^{-1} were retained. Heavily salvaged stands were located in lands designated as matrix, and 5–8 snags ≥ 50.8 cm DBH ha^{-1} were retained. Allocation of treatments to stands was completed by staff of the Deschutes National Forest prior to the beginning of our study.

2.2. Study stand selection

We restricted our selection of stands to those having similar aspect (90 – 180°) and elevation (1350–2025 m), moderate or high burn intensity, a density of ≥ 9 trees ≥ 91.4 cm DBH stand^{-1} prior to the fire, an area of at least 7 ha, and a shape allowing placement of at least three non-overlapping 80 m radius circles for bird sampling. Only four stands of each treatment met these criteria, and we used all of these as study stands ($n = 12$). Selected stands were located in areas of high burn intensity, dominated by standing dead trees (snags). The few live trees that were present in some sites were primarily along stand edges. Study stands ranged from 10 to 112 ha in size (unsalvaged stands: 10–18 ha, moderately salvaged stands: 20–112 ha, and heavily salvaged stands: 13–45 ha). We selected three points in each stand for bird sampling with each point separated by ≥ 160 m in all stands and ≥ 100 m from stand edges where possible. In stands of area >40 ha, we clustered the points to mimic the layout in smaller sized stands.

Salvage treatments were implemented prior to the beginning of our study. Secondary fuel treatments, to decrease the risk of future fire, occurred between out two field seasons in seven stands (all of the heavily salvaged stands and three of the moderately salvaged stands; one moderately salvaged stand did not warrant treatment because of little standing and on the ground fuel). Fuels treatments involved felling snags <20.3 cm DBH, cutting them into 4.3 m lengths. One moderate salvage stand was broadcast burned in the fall of 2005.

2.3. Vegetation sampling

We established four vegetation plots at each bird sampling point. One plot was centered on the bird sampling point and three satellite plots were located 30 m from the center point and radially spaced 120° from one another; the compass bearing of the first satellite point was randomly selected. Each vegetation plot consisted of nested 5 and 11.3 m radii circles. We modified the BBIRD vegetation protocols (Martin et al., 1997), and visually estimated the percent ground cover of grass, forbs, shrubs, downed wood (≥ 10 cm in diameter), small wood (<10 cm in diameter), and bare soil within the 5 m radius circle. Within the 11.3 m radius circle, we measured DBH of all standing snags ≥ 10 cm DBH and ≥ 2 m tall. We tallied the number of snags <10 cm DBH and ≥ 2 m tall. We divided all snags into five DBH categories (≤ 20 , >21 – 35 , >36 – 50 , >51 – 90 , and >91 cm) and examined the diameter distribution among treatments. We compared densities of snags >35 cm DBH among treatments and years; we selected this cut-off given the implemented salvage prescription and the preference of wildlife for large snags.

At each plot, we established two transects to measure volume of downed wood and shrubs. We established transects at bearings $\pm 45^\circ$ of the plot's aspect. Transects were 22.6 m long and intersected at their midpoints at the center of the vegetation plot. We identified and tallied all shrubs ≥ 0.25 m tall that intersected the transect for ≥ 0.05 m. We measured height, and maximum widths parallel and perpendicular to the transect for each shrub. We recorded a shrub twice if it intersected both transects. We calculated volume of shrubs as:

$$\text{Shrub volume} = \frac{4}{3}\pi \frac{w_1 w_2 h}{2} \text{ (m}^3 \text{ ha}^{-1}\text{)}$$

where w_1 = parallel width (m), w_2 = perpendicular width (m), and h = height (m).

We adapted Waddell's (2002) protocol for measuring downed wood. Downed wood was measured if (1) it was ≥ 10 cm in diameter where crossed by the transect; (2) the central longitudinal axis intersected the transect; (3) the decay class was ≤ 4 (Cline et al., 1980); (4) it no longer had roots in the ground; and (5) it was not elevated $>45^\circ$ above the ground. We recorded the diameter of downed wood at large and small ends (minimum 10 cm), and overall length (to 10 cm diameter). We calculated volume of downed wood using the formula:

$$\text{volume}_m = \frac{(\pi \times 8)(D_s^2 + D_l^2) \times l}{10000} \text{ (m}^3 \text{ ha}^{-1}\text{)}$$

where D_s = diameter of small end (cm), D_l = diameter of large end (cm), and l = length of downed wood (m). We measured diameter and length to the first clean break of the piece of downed wood. Downed wood was recorded twice if it intersected both transects (Waddell, 2002).

2.4. Bird sampling

We counted birds between one half-hour before and 4 h after sunrise, on days without heavy rain, fog, snow, or wind between 20 May and 5 July 2005 and 2006. We recorded all birds seen or heard at each point within 8 min, and used a range finder and flagging placed at 25 m intervals to assist in estimating the distance from the observer to each bird (Ramsey and Scott, 1979; Reynolds et al., 1980). We determined sampling order within a site by randomly selecting the first of three points and ordered subsequent points to maximize logistic efficiency. We sampled two stands each morning, with all 12 stands sampled before any were resampled. Each stand was sampled five times each year.

2.5. Density and relative abundance estimates

To account for potential variation in detectability of species (Buckland et al., 2001), we used multiple covariate distance sampling (MCDS) in program DISTANCE 5.0 (Thomas et al., 2006) to estimate densities (birds ha^{-1}) of all species with >85 detections and ≥ 1 detection in each treatment. Six species and one genus satisfied these criteria. We analyzed the violet-green swallow (*Tachycineta thalassina*) and tree swallow (*T. bicolor*) together because we were unable to reliably identify all individuals to species during point counts. We are aware of the potential problems of estimating density as pointed out by Johnson (2008). We attempted to minimize violation of assumptions by training observers before beginning point counts and using a range finder to aid in estimation of distances. We followed general guidelines outlined in Buckland et al. (2001) for our analyses. Before fitting a detection probability, we pooled all data for a species and truncated the data at the distance where the probability of detection fell below 0.1. We included a generic multiplier ($n = 1/5$) to account for multiple visits in our density estimates. We grouped the data into distance intervals if necessary based on visual inspections of detection probability plots and chi-goodness of fit. We selected a hazard-rate detection function with a cosine expansion series for each species based on this model having the smallest AIC_c value (Akaike's information criterion corrected for small sample sizes). To assess constancy of detection probabilities across years, salvage treatments, and observers, we modeled detection probabilities for each species by stratum (salvage treatment by year), all detections (global), and all detections with one of three potential covariates (year, salvage treatment, and observer) using MCDS (Buckland et al., 2004). We calculated ΔAIC_c as the difference between the AIC_c value for a given model and the model with the lowest AIC_c value for each

species (Buckland et al., 2004). For species with models that were stratified by treatment \times year or that included a covariate having $\Delta AIC_c \leq 2.0$, we modeled detection probability as a function of salvage treatment or year independently. We selected the final model using the lowest AIC_c value.

For nine species that we did not detect frequently enough to estimate density (≥ 20 total detections), we calculated relative abundance as:

$$RA_{ijk} = \frac{D_{ijk}}{V_{jk}}$$

where RA_{ijk} = relative abundance of species i in stand j and year k , D_{ijk} = number of detections of species i in stand j and year k , and V_{jk} = number of visits to stand j in year k . We used all detections ≤ 75 m from the sampling point for all of these species except black-backed woodpeckers; we extended the radius to 100 m for black-backed woodpeckers. Probabilities of detection 0.80 at distances >75 m for all species except for the hairy fell below woodpecker, where probabilities of detection fell below 0.80 at distances >100 m. We used these distances as cut points to approximate equal detectability of species among treatments.

2.6. Statistical analysis

We used SAS 9.1.3 for all statistical tests (SAS Institute Inc., 2004). We used a generalized linear mixed model with repeated measures on year (PROC MIXED) to determine if vegetation structure (diameter of snags ≥ 10 cm, number of snags >35 cm DBH, shrub volume, volume of downed wood) and percent cover differed among salvage treatments or between years. We considered salvage treatment, year, and the interaction of salvage treatment and year as fixed effects, and stand variation within treatments as a random effect. Our model allowed variance to differ between years. We selected a covariance structure from four possible structures (compound symmetry, unstructured covariance [1] or [2], and autoregressive covariance) and used the structure with the lowest AIC_c value. We ln-transformed shrub volume to better meet model assumptions of constant variance and normal distribution of residuals. If the treatment effect was significant, we used a Tukey test (Zar, 1999) to assess all three possible pair-wise comparisons.

We also used the analytic structure described above to determine if density or relative abundance of each species differed among salvage treatments or between years (PROC MIXED). Standard errors associated with estimates of densities in each stand varied depending on the number of observations of each species. To account for this variation, we weighted densities by the inverse of the standard error, giving more weight to estimates with more confidence (Ramsey and Schafer, 2002). We tested for significance ($\alpha = 0.05$) of the fixed effects before evaluating five *a priori* hypotheses regarding density or relative abundance of each species.

We evaluated orthogonal contrasts using ANOVA results to assess three *a priori* hypotheses regarding the influence of salvage. Our contrasts examined the differences between estimates for (a) unsalvaged treatment (U) and the average of moderate and heavy salvage treatments $((M+H)/2)$, and (b) the moderately salvaged treatment (M) and heavily salvaged treatment (H) for each species. Our hypotheses were: (1) the null treatment effect hypothesis—no differences in abundances among salvage treatments ($U = (M+H)/2$; $M = H$); (2) the salvage-effect hypothesis—abundances differ between the unsalvaged treatment and the moderate and heavy salvage treatments ($U \neq (M+H)/2$), but are similar between salvage intensities ($M = H$); and (3) the graded-response hypothesis—abundances differ between the unsalvaged treatment and the moderate and heavy salvage treatments ($U \neq (M+H)/2$), and abundances differ between the moderate salvage and heavy

salvage intensities ($M \neq H$). A fourth response is also possible; abundances could be similar between the unsalvaged treatment and moderate and heavy treatments ($U = (M+H)/2$), but differ between the moderate and heavy salvage treatment ($M \neq H$). Although conceivable, this does not correspond to any *a priori* hypothesis, and no species responded in a manner consistent with this pattern. We also evaluated two *a priori* hypotheses regarding year effect for each species: (4) the null temporal effect hypothesis—abundances did not differ between years (2005 = 2006); and (5) the year-effect hypothesis—abundances are different between years (2005 \neq 2006).

We ln-transformed densities before analysis to better meet the assumption of normally distributed residuals. Back-transformation of the differences among fixed effects yielded an estimate of the ratio of differences. We conducted a retrospective power analysis (Sokal and Rohlf, 1995) to estimate the percent difference (10–90% in 10% increments) in densities or relative abundances of birds among the treatments in order to have $\geq 80\%$ power, given the sample size ($n = 4$). We calculated coefficients of variation (Sokal and Rohlf, 1995) from this and other studies (Hagar et al., 1996; Schulte and Niemi, 1998; Smucker et al., 2005; Sallabanks et al., 2006) to estimate variability of abundance estimates for each species of bird. We used one-sided tests of the null hypothesis that densities and relative abundances did not differ among treatments of salvage ($\alpha = 0.05$).

We used multiple linear regression to model the density or relative abundance of each species of bird as a discrete function of vegetation (mean volume of shrubs, mean volume of downed wood, and mean number of snags > 35 cm DBH) at the stand level to determine if abundances of species were related to vegetation characteristics (PROC REG). We tested for correlation among variables before using them as predictors of densities of birds. If $|r|$ was ≥ 0.5 between two variables, we interpreted only one of the variables, but reported all coefficients. We ln-transformed the variables shrub volume and downed wood volume to meet the model assumptions of normal distribution and constant variance of the residuals, as there was an order of magnitude difference among values for both variables.

3. Results

3.1. Vegetation

The diameter distribution of snags ≥ 2 m tall changed in the moderate and heavy salvage treatments between 2005 and 2006,

primarily because of removal of snags < 20 cm DBH in fuel treatments (Fig. 1). Differences in mean diameter of snags ≥ 10 cm DBH among salvage treatments depended on year ($F_{2,9} = 16.77$, $P = 0.0009$; Fig. 2). Mean diameter of snags ≥ 10 cm DBH did not differ among salvage intensities in 2005 ($F_{2,9} = 0.25$, $P = 0.78$), but was 12.95 cm smaller (95% confidence interval [CI]: 1.42–24.29) in the unsalvaged treatment than in the heavy salvage treatment in 2006. The mean diameter of snags ≥ 10 cm DBH increased by 9.36 cm (95% CI: 3.45–15.28) in the moderate salvage treatment and by 13.7 cm (95% CI: 7.79–19.61) in the heavy salvage treatments from 2005 to 2006.

Mean density of snags > 35 cm DBH did not differ between 2005 and 2006 ($F_{1,9} = 0.06$, $P = 0.81$; Fig. 2). The unsalvaged treatment had significantly greater densities of snags > 35 cm DBH than did the moderate (\bar{x} difference = 92.2 snags ha^{-1} , 95% CI: 53.6–130.8, $t_9 = 6.74$, $P = 0.0002$) and the heavy (\bar{x} difference = 111.4 snags ha^{-1} , 95% CI: 72.9–150.0, $t_9 = 8.1$, $P < 0.92$) salvage treatment in 2005 and 2006 (Fig. 2). Although point estimates for densities of snags > 35 cm DBH were greater in moderately salvaged stands than in heavily salvaged stands (Fig. 2), densities did not statistically differ significantly ($t_9 = 1.46$, $P = 0.40$) because of high spatial variability in distribution of snags and the inability of our sampling techniques to fully capture stand-level differences using our plot-based sampling.

Shrub volume increased across all treatments between 2005 and 2006, though the magnitude of the difference depended on the salvage treatment ($F_{2,9} = 6.48$, $P = 0.018$; Fig. 2). Greatest volume of shrubs occurred in the heavily salvaged treatment in 2006 (112.42 $\text{m}^3 \text{ha}^{-1}$), and the least volume of shrubs occurred in the unsalvaged treatment in 2005 (2.25 $\text{m}^3 \text{ha}^{-1}$). Volume of downed wood increased significantly in moderately salvaged and unsalvaged treatments between 2005 and 2006 (moderate: $F_{1,9} = 12.24$, $P = 0.0067$; unsalvaged: $F_{1,9} = 9.55$, $P = 0.013$; Fig. 2).

Bare ground was significantly greater in unsalvaged than in moderate (\bar{x} difference = 15.59%, 95% CI: 2.49–28.68, $t_9 = 3.32$, $P = 0.009$; Table 1) and heavy salvage treatments (\bar{x} difference = 13.45%, 95% CI: 0.36–26.55, $t_9 = 2.87$, $P = 0.045$). Bare ground decreased (\bar{x} difference = 15.72%, 95% CI: 10.91–20.53, $F_{1,9} = 54.71$, $P < 0.0001$), and cover of shrubs (\bar{x} increase = 15.74%, 95% CI: 9.52–21.96, $F_{1,9} = 31.79$, $P = 0.0003$), and cover of downed wood (\bar{x} increase = 3.80%, 95% CI: 1.71–5.81, $F_{1,9} = 16.89$, $P = 0.003$) increased significantly among all treatments between 2005 and 2006. Cover of shrubs and downed wood did not significantly differ among treatments. Cover of small wood (< 10 diameter) was significantly

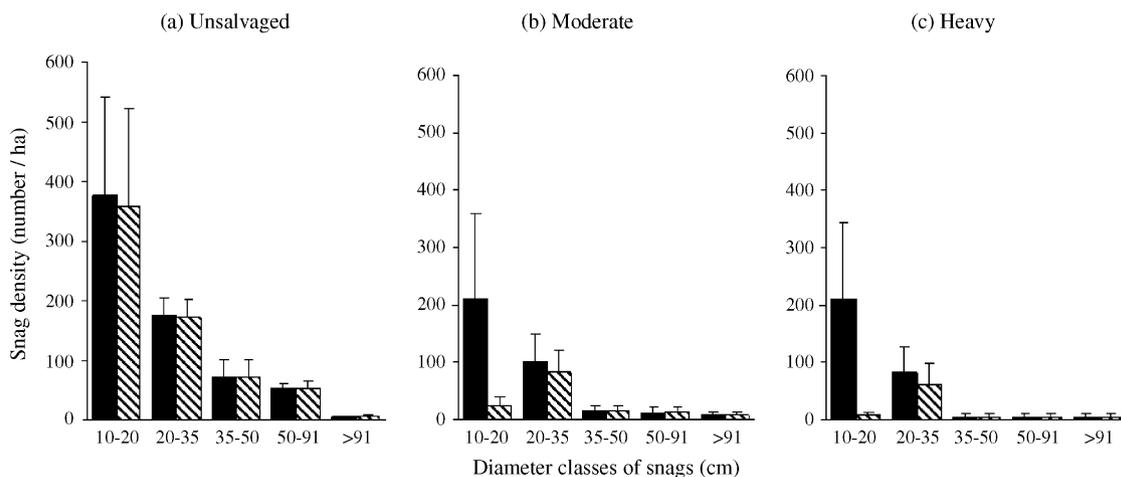


Fig. 1. Diameter distribution of snags in postfire forests with three treatments of salvage logging ($n = 4$ stands treatment $^{-1}$) in Deschutes National Forest, OR, summers 2005–2006. Mean estimates for 2005 (black) and 2006 (hatched); error bars represent 95% confidence limits. Minimum height of snag to be counted was 2 m. Decrease of snags < 20 cm dbh between 2005 and 2006 in moderate and heavy salvage treatment is primarily due to implementation of fuels reduction treatment. Unsalvaged treatment had greater densities of snags > 35 cm dbh than moderate or heavy salvage treatment.

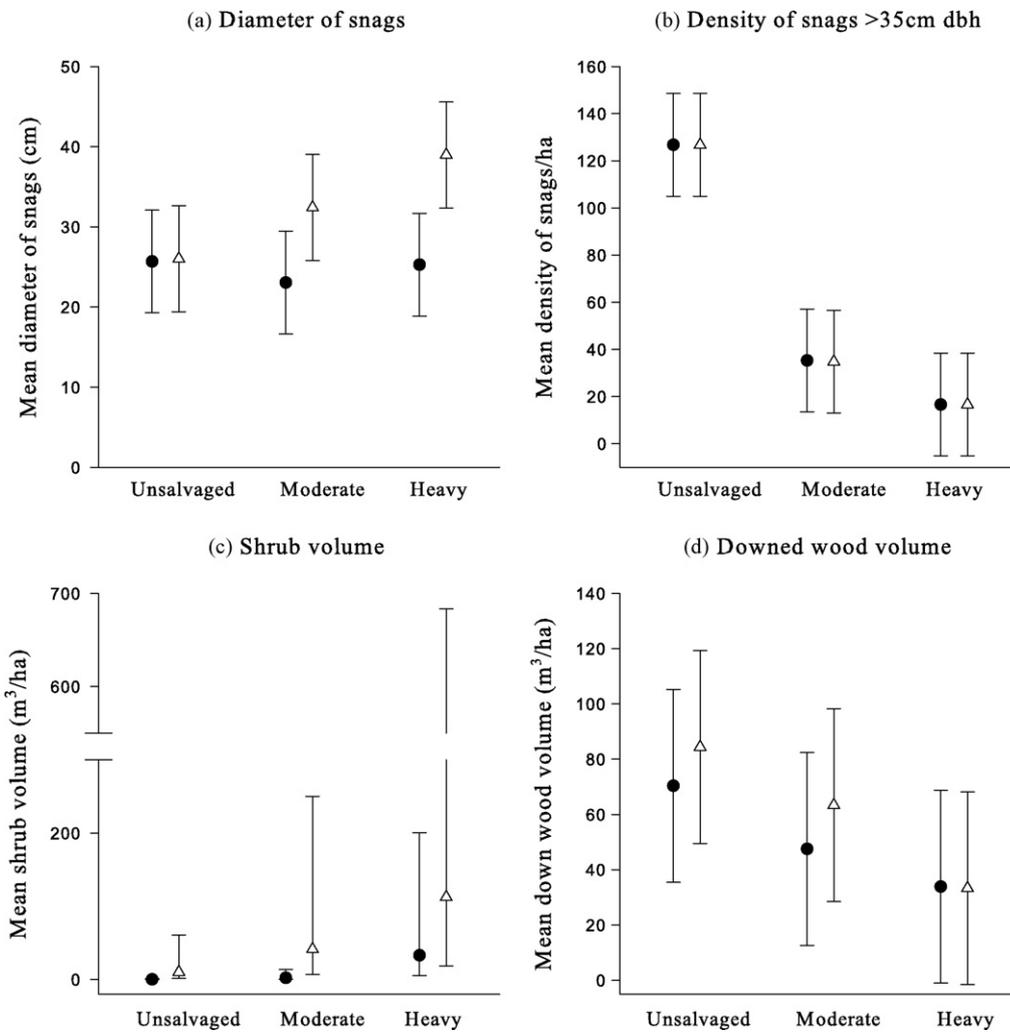


Fig. 2. Mean or median estimates of vegetation variables in postfire forests with three treatments of salvage logging ($n = 4$ stands treatment⁻¹) in Deschutes National Forest, OR, summers 2005–2006. Point estimates for 2005 (●) and 2006 (△); error bars represent 95% confidence limits.

less in the unsalvaged than in the moderate (\bar{x} difference = 13.92%, 95% CI: 7.97–19.88, $t_9 = 6.53$, $P = 0.0003$) and heavy (\bar{x} difference = 10.64%, 95% CI: 4.68–16.59, $t_9 = 4.99$, $P = 0.002$) salvage treatments, and did not differ between years ($F_{1,9} = 1.06$, $P = 0.33$). Cover of forbs ($F_{2,9} = 0.58$, $P = 0.58$) and grasses ($F_{2,9} = 0.23$, $P = 0.80$)

did not differ significantly among treatments. Grasses and forbs were not a dominant ground cover at our study site, and while we detected a statistically significant increase in grass between years (\bar{x} increase = 0.65%, 95% CI: 0.33–0.97%, $F_{1,9} = 20.92$, $P = 0.0013$), the biological significance is negligible.

Table 1

Mean percent cover of vegetation in postfire forests with three treatments of salvage logging ($n = 4$ stands treatment⁻¹) in Deschutes National Forest, OR, summers 2005–2006. Estimates for unsalvaged (U), moderate salvage intensity (M), and heavy salvage intensity (H) with 95% confidence limits in parentheses. Bare ground is the amount of exposed soil; shrub, grass, and forbs variables combined all species; downed wood (≥ 10 cm dbh) and small wood (< 10 cm dbh) included material left after salvage logging operations.

Variable	2005			2006		
	U	M	H	U	M	H
Bare ground	60.11 (51.01, 69.20)	39.79 (30.69, 48.89)	45.31 (36.22, 54.41)	40.34 (32.30, 48.37)	29.48 (21.44, 37.51)	28.23 (20.19, 36.26)
Shrub	13.73 (3.96, 23.50)	13.02 (3.25, 22.79)	15.83 (6.06, 25.61)	31.75 (20.53, 42.97)	29.4 (18.17, 40.62)	28.67 (17.45, 39.89)
Downed wood	5.75 (1.69, 9.81)	12.36 (8.29, 16.42)	8.46 (4.40, 12.52)	6.91 (2.03, 11.78)	15.31 (10.43, 20.19)	15.75 (10.87, 20.63)
Small wood	4.62 (-0.40, 9.63)	23.5 (18.49, 28.51)	16.53 (11.52, 21.55)	7.29 (3.51, 11.07)	16.25 (12.47, 20.03)	16.65 (12.87, 20.42)
Grass	1.02 (-0.12, 2.16)	1.19 (0.05, 2.33)	0.75 (-0.39, 1.89)	1.52 (0.38, 2.66)	1.94 (0.80, 3.08)	1.44 (0.30, 2.58)
Forbs	2.33 (0.92, 3.73)	0.92 (-0.49, 2.32)	0.89 (-0.52, 2.29)	0.74 (-0.68, 2.15)	0.76 (-0.65, 2.17)	0.78 (-0.62, 2.19)

Table 2
ANOVA of mean relative abundances (R) and median density estimates (D) as a function of fixed effects in a postfire forest with three treatments of salvage logging ($n = 4$ stands treatment⁻¹) in Deschutes National Forest, OR, summers 2005–2006.

Species	Fixed effects					
	Year		Treatment		Year × treatment	
	F _{1,9}	P	F _{2,9}	P	F _{2,9}	P
Hairy woodpecker (D)	0.07	0.79	4.69	0.040	1.19	0.35
Black-backed woodpecker (R)	3.66	0.08	10.18	0.005	3.39	0.08
Swallows (D)	17.94	0.002	1.26	0.33	0.57	0.59
Western wood-pewee (R)	1.00	0.34	4.38	0.047	2.33	0.15
Dusky flycatcher (R)	1.00	0.34	2.38	0.15	1.28	0.33
Brown creeper (R)	3.27	0.10	9.62	0.006	1.16	0.37
White-breasted nuthatch (R)	0.26	0.62	0.22	0.80	0.12	0.89
Red-breasted nuthatch (R)	22.01	0.001	0.62	0.56	1.81	0.22
House wren (D)	0.12	0.74	0.38	0.69	0.78	0.49
Mountain bluebird (D)	0.55	0.48	1.27	0.33	0.44	0.66
American robin (R)	0.74	0.41	2.50	0.14	0.83	0.47
Yellow-rumped warbler (D)	7.42	0.024	9.69	0.006	0.47	0.64
Western tanager (R)	1.62	0.24	0.63	0.56	0.63	0.56
Chipping sparrow (R)	2.21	0.17	0.50	0.62	0.08	0.93
Fox sparrow (D)	5.06	0.051	6.26	0.020	0.41	0.67
Dark-eyed junco (D)	65.6	<0.0001	5.09	0.033	1.17	0.35

Density of snags >35 cm DBH positively correlated with volume of downed wood ($r = 0.54$, $P = 0.006$), and negatively correlated with shrub volume ($r = -0.61$, $P = 0.002$). Downed wood volume negatively correlated with shrub volume ($r = -0.553$, $P = 0.005$).

3.2. Birds

We detected 34 species of birds, of which 29 species were detected both years (Cahall, 2007). We estimated relative abundance (birds stand⁻¹) of nine species (black-backed woodpecker, *Picoides arcticus*; western wood-pewee, *Contopus sordidulus*; dusky flycatcher, *Empidonax oberholseri*; brown creeper, *Certhia americana*; white-breasted nuthatch, *Sitta carolinensis*; red-breasted nuthatch, *Sitta canadensis*; American robin, *Turdus migratorius*; western tanager, *Piranga ludoviciana*; and chipping sparrow, *Spizella passerina*; Appendix A) and density (birds ha⁻¹) of six species and one genus (hairy woodpecker, *P. villosus*; tree and violet-green swallows, *Tachycineta* species; house wren, *Troglodytes aedon*; mountain bluebird, *Sialia currucoides*; yellow-rumped warbler, *Dendroica coronata*; fox sparrow, *Passerella iliaca*; and

dark-eyed junco, *Junco hyemalis*; Appendix B) for each treatment of salvage and for each year. Our results from analyses using estimates of density were qualitatively similar to results using relative abundance for each taxon. A treatment by year interaction was not significant for any taxon ($P \geq 0.08$; Table 2). Of the 16 taxa for which we estimated relative abundance or density, nine did not significantly differ among treatments (swallows, dusky flycatcher, white-breasted nuthatch, red-breasted nuthatch, house wren, mountain bluebird, American robin, western tanager, and chipping sparrow; Table 2). A retrospective power analysis for these species indicated that power to detect differences among the orthogonal contrasts varied by species. For seven species (dusky flycatcher, white-breasted nuthatch, red-breasted nuthatch, mountain bluebird, American robin, western tanager, and chipping sparrow) differences in abundances among treatments had to be $\geq 30\%$ in order for power to be $\geq 80\%$. It is possible that differences in relative abundances or densities $< 30\%$ could be of biological importance for some of these species, and as a result a conservative approach is to neither reject nor accept the null hypothesis of no difference among treatments. However, our results suggest any differences

Table 3
Estimates of differences of mean relative abundances (R) and median densities (D) from orthogonal contrasts among three treatments of salvage logging ($n = 4$ stands treatment⁻¹) in a postfire forest, Deschutes National Forest, OR, summers 2005–2006. Comparisons are unsalvaged to average of moderate and heavy salvage (U–(M+H)/2), and moderate salvage to heavy salvage (M–H). Confidence limits that include 0 (1) indicate no statistical significance of differences in relative abundances (densities) among treatments ($\alpha = 0.05$).

Species	Comparison					
	U–(M+H)/2			M–H		
	Difference	95% CI	P	Difference	95% CI	P
Hairy woodpecker (D)	1.258	1.034–1.530	0.027	0.875	0.712–1.075	0.18
Black-backed woodpecker (R)	0.675	0.320–1.030	0.002	–0.250	–0.660–0.160	0.20
Swallows (D)	1.150	0.832–1.590	0.36	0.827	0.573–1.194	0.27
Western wood-pewee (R)	0.288	0.067–0.508	0.016	–0.025	–0.279–0.229	0.83
Dusky flycatcher (R)	–0.363	–1.001–0.276	0.23	–0.575	–1.312–0.162	0.11
Brown creeper (R)	0.888	0.426–1.349	0.002	–0.125	–0.658–0.408	0.61
White-breasted nuthatch (R)	–0.100	–0.468–0.268	0.55	–0.050	–0.475–0.375	0.80
Red-breasted nuthatch (R)	0.000	–0.264–0.264	1.00	–0.150	–0.455–0.155	0.29
House wren (D)	0.885	0.644–1.216	0.41	0.985	0.675–1.437	0.93
Mountain bluebird (D)	0.754	0.504–1.127	0.15	1.020	0.629–1.655	0.93
American robin (R)	0.225	–0.007–0.457	0.68	0.050	–0.281–0.318	0.06
Yellow-rumped warbler (D)	2.078	1.420–3.043	0.002	0.901	0.619–1.313	0.55
Western tanager (R)	–0.013	–0.233–0.208	0.90	–0.125	–0.379–0.129	0.30
Chipping sparrow (R)	0.275	–0.379–0.929	0.93	–0.100	–0.855–0.655	0.66
Fox sparrow (D)	0.654	0.467–0.916	0.019	0.588	0.371–0.932	0.029
Dark-eyed junco (D)	0.737	0.591–0.920	0.012	1.090	0.832–1.428	0.49

Table 4

Pearson's correlation (r) coefficients between density (D) or relative abundance (R) of species of birds and vegetation variables in 12 postfire forest stands with three treatments of salvage logging, Deschutes National Forest, OR, summers 2005–2006 ($n = 24$).

Species	Vegetation variables		
	Shrub volume [†]	Down wood volume [†]	Number of snags >35 cm ha ⁻¹
Hairy woodpecker (D)	-0.11	0.00	0.40
Black-backed woodpecker (R)	-0.14	0.12	0.51
Swallows (D)	0.33	-0.01	0.14
Western wood-pewee (R)	-0.27	0.44	0.65
Dusky flycatcher (R)	0.45	-0.47	-0.31
Brown creeper (R)	-0.54	0.15	0.64
White-breasted nuthatch (R)	0.12	-0.34	-0.11
Red-breasted nuthatch (R)	-0.30	0.10	-0.03
House wren (D)	0.31	-0.19	-0.17
Mountain bluebird (D)	0.23	-0.12	-0.27
American robin (R)	-0.26	0.13	0.42
Yellow-rumped warbler (D)	-0.56	0.25	0.71
Western tanager (R)	-0.01	-0.11	-0.02
Chipping sparrow (R)	0.29	-0.21	0.13
Fox sparrow [†] (D)	0.81	-0.65	-0.64
Dark-eyed junco (D)	0.60	-0.03	-0.30

[†] Some variables were ln-transformed to better meet model assumptions.

among treatments are not likely to be large for these seven species. For house wrens and swallows, differences in abundances $\geq 10\%$ among treatments could be detected with $\geq 80\%$ power, and as a consequence we conclude that biologically significant differences in abundances were not likely among treatments for these taxa.

Densities of the hairy woodpecker and yellow-rumped warbler and relative abundances of the black-backed woodpecker, western wood-pewee, and brown creeper were greater in the unsalvaged treatment than in salvaged treatments, but did not differ between salvage intensities (Table 3), consistent with the salvage effect hypothesis. Densities of the dark-eyed junco were greater in salvaged than unsalvaged stands (Table 3), also consistent with the salvage effect hypothesis. Densities of the fox sparrow were highest in the heavily salvaged stands, intermediate in the moderately salvaged stands, and lowest in the unsalvaged stands (Table 3, Appendix B) exhibiting a pattern consistent with the graded response hypothesis.

In 2006, densities of the dark-eyed junco ($\bar{x} = 1.68$ times greater, 95% CI: 1.45–1.94) and swallows ($\bar{x} = 1.61$ times greater, 95% CI: 1.25–2.09) were greater than in 2005. Density of the yellow-rumped warbler ($\bar{x} = 0.72$ times lower, 95% CI: 0.54–0.95) and relative abundance of the red-breasted nuthatches (\bar{x} decrease = 0.52 birds stand⁻¹, 95% CI: 0.27–0.77) decreased significantly between 2005 and 2006 (Table 2).

Densities of the yellow-rumped warbler ($r = 0.71$) and relative abundances of the black-backed woodpecker ($r = 0.51$), brown creeper ($r = 0.64$), western wood pewee ($r = 0.65$), and American robin ($r = 0.42$) were positively correlated with density of snags >35 cm DBH (Table 4). Densities of the fox sparrow ($r = 0.80$) and dark-eyed junco ($r = 0.60$) and relative abundances of the dusky flycatcher ($r = 0.45$) were positively correlated with shrub volume (Table 4). Densities or relative abundances of eight taxa (hairy woodpecker, white-breasted nuthatch, red-breasted nuthatch, house wren, mountain bluebird, western tanager, chipping sparrow, and swallows) were not strongly correlated with any of the vegetation variables measured ($|r| < 0.40$; Table 4).

4. Discussion

4.1. Salvage-effect hypothesis

Patterns for six species (black-backed woodpecker, hairy woodpecker, brown creeper, western wood-pewee, yellow-rumped warbler, and dark-eyed junco) were consistent with

predictions of the salvage-effect hypothesis, with species abundances differing between the unsalvaged and salvage treatments. For these species, the moderate salvage intensity did not mitigate differences in bird density or abundance in the unsalvaged treatment compared to the more intense salvage treatment. Our findings are similar to a majority of other studies that have examined the response of woodpeckers to postfire salvage logging (e.g., Saab and Dudley, 1998), and provide information on the response of cup-nesting species, of which there was little information.

Greater abundances of black-backed and hairy woodpeckers in the unsalvaged treatment relative to the salvaged treatments, even though both species of woodpeckers have larger territories than our study stands, suggests that small patches of unsalvaged forest can influence abundances. The black-backed woodpecker is closely associated with burned forests, and abundances typically dramatically increase after fire (Blackford, 1955; Hutto, 1995; Murphy and Lehnhausen, 1998; Dixon and Saab, 2000), and as a consequence are of particular conservation concern (USDI, 2003; Oregon Natural Heritage Information Center, 2004; Montana Natural Heritage Program and Montana Fish Wildlife and Parks, 2006). Our findings of lower numbers in salvaged stands are consistent with those of other studies that have demonstrated negative effects of salvage logging on the black-backed woodpecker (Caton, 1996; Hutto and Gallo, 2006; Koivula and Schmiegelow, 2007; Saab et al., 2007), even in partially salvaged stands (Saab and Dudley, 1998; Haggard and Gaines, 2001). Similarly, numbers of the hairy woodpecker appear to be negatively influenced by salvage logging at both intensities considered in our study, consistent with findings in other regions where near complete removal of snags occurred (Caton, 1996; Saab and Dudley, 1998; Haggard and Gaines, 2001; Hutto and Gallo, 2006; Koivula and Schmiegelow, 2007; Saab et al., 2007). Yet, findings for response of the hairy woodpecker to partial salvage logging relative to unsalvaged and complete salvage have been mixed (Saab and Dudley, 1998; Haggard and Gaines, 2001; Saab et al., 2007), suggesting that further investigation of different salvage intensities are necessary to better understand the response of the hairy woodpecker.

Our findings were qualitatively similar to those of other studies that have examined responses of the brown creeper, western wood-pewee, and yellow-rumped warbler to postfire salvage logging (Haggard and Gaines, 2001; Morissette et al., 2002; Fontaine, 2007). The findings of Fontaine (2007) further suggest

that the yellow-rumped warbler is positively associated with edges of salvage-logged forests.

Two previous studies examined the response of the dark-eyed junco to postfire salvage logging and found different responses in mixed evergreen (Fontaine, 2007) and boreal forest types (Morissette et al., 2002). The dark-eyed junco occurred more frequently in salvaged mixed-evergreen and mixed-wood boreal forests than in unsalvaged stands, but occurred less frequently in salvaged than in unsalvaged jack pine forests (Morissette et al., 2002; Fontaine, 2007). The strong positive correlation between the dark-eyed junco and shrub volume suggests they may be responding to fire effects on shrubs and not snag density. Still, causal mechanisms underlying region- and habitat-specific differences are unclear.

4.2. Graded-response hypothesis

Of the species we examined, only the fox sparrow demonstrated patterns consistent with the graded-response hypothesis, which predicts that abundances differ between the unsalvaged and the salvaged stands, and between moderately and heavily salvaged stands. Densities of the fox sparrow were strongly positively correlated with volume of shrubs which increased with decreasing density of snags. The fox sparrow selects brushy habitat (Austin, 1968; Weckstein et al., 2002), and densities of the fox sparrow increase as shrub density increases following fire (Bock and Lynch, 1970; Raphael et al., 1987). Changes in shrub volume in the salvaged stands may have been an underlying mechanism for observed differences in densities of the fox sparrow. To our knowledge, abundances of the fox sparrow in salvage-logged habitat after fire have not been documented previously.

4.3. Null treatment effect

We were unable to detect statistically significant differences in densities or relative abundances among treatments for nine taxa (swallows, dusky flycatcher, white-breasted nuthatch, red-breasted nuthatch, house wren, mountain bluebird, American robin, western tanager, and chipping sparrow). Demonstrating no effect is, at best, methodologically challenging (Steidl et al., 1997). Our retrospective power analysis indicates that biologically significant differences in numbers of swallows and house wrens among treatments were unlikely. For seven species (American robin, chipping sparrow, dusky flycatcher, mountain bluebird, red-breasted nuthatch, white-breasted nuthatch, and western tanager) our study had a statistical power <0.8 to detect differences in abundances <30% among treatments. Thus, while relatively large (>30%) differences among treatments are unlikely for these species, differences smaller than this are possible and may not have been detected using our experimental design. Determining what level of difference is of biological importance is problematic and often subjective, but we contend that our data indicate that the likelihood of large biologically significant differences is small for these seven species.

Five of the taxa for which we were unable to detect statistically significant differences among treatments are cavity-nesters (swallows, white-breasted nuthatch, red-breasted nuthatch, house wren, and mountain bluebird) that have demonstrated mixed responses to fire (Bock and Lynch, 1970; Raphael and White, 1984; Hejl et al., 1995; Kotliar et al., 2002; Bock and Block, 2005; Hannon and Drapeau, 2005; Saab et al., 2005; Kirkpatrick et al., 2006) or postfire salvage logging (Haggard and Gaines, 2001; Morissette et al., 2002; Hutto and Gallo, 2006). Three cup-nesting species of birds (American robin, chipping sparrow, and western tanager) have also been reported to demonstrate mixed responses to fire (Hutto, 1995; Schulte and Niemi, 1998; Kotliar et al., 2002; Hannon

and Drapeau, 2005; Saab et al., 2005) and postfire salvage logging (LeCoure et al., 2000; Morissette et al., 2002; Schwab et al., 2006). Variation in salvage prescriptions, treatment implementation, fire severity, and time since fire (Smucker et al., 2005) may influence the response of species to salvage logging, as well as regional and habitat differences for each species in the previous studies. Further, these species may require specific habitat components, such as open habitat or shrubs for foraging, which vary in ways independent of salvage treatments across regions.

4.4. Temporal patterns

We detected temporal changes in densities or relative abundances of four taxa; swallows and the dark-eyed junco increased and the yellow-rumped warbler and red-breasted nuthatch decreased from 2005 to 2006 (Table 2). It is difficult to separate local temporal changes in numbers from regional or broad-scale changes in populations, particularly in a short-term study. Although bird community changes are associated with habitat changes through time (James and Warner, 1982), vegetation and habitat changes may not have differed adequately from the initial conditions to cause measurable differences in abundance for most of the species in our study. Persistence of these early patterns (increasing or decreasing abundances) may be of particular interest as forest succession proceeds.

4.5. Scope and limitations

There are several factors of our study design that may have influenced our results. First, our inability to randomly allocate treatments and collect pre-salvage data does not allow us to infer causality for observed differences among treatments of salvage logging. Therefore, it is possible that observed patterns of bird abundance could be due to some factor related to pre-existing differences in habitat condition rather than salvage logging. However, we are unaware of any differences in pre-treatment conditions that would result in the observed differences. This and the correspondence of our findings for some species (notably brown creeper, black-backed woodpecker and hairy woodpecker) with those of other studies (Saab and Dudley, 1998; Haggard and Gaines, 2001; Hutto and Gallo, 2006; Saab et al., 2007) suggest our findings are likely a result of salvage logging. Second, the small size of our study stands, particularly for the unsalvaged treatment, makes generalizations to larger stands difficult. Undoubtedly the responses of some species to patches of unsalvaged forest in a matrix of salvaged stands differ from what might occur in larger unsalvaged tracts. Moreover, territories of some species (e.g., woodpeckers) likely extended beyond the area of our study stands. Nevertheless, the sizes of our stands reflect the operational scale of management in some forest systems, and our results are highly relevant to activities occurring at that scale. Third, the retention of large snags (>35.6 cm DBH) in our study likely influenced the densities and relative abundances of bird species that we observed. As densities and characteristics of snags following disturbance can strongly influence communities and abundance of birds (Lehmkuhl et al., 2003; Stuart-Smith et al., 2006), our results are unlikely to be applicable to salvage practices that do not retain large diameter trees. Fourth, our results pertain to the early (<3 years post-disturbance) years after stand-replacing fire and subsequent salvage logging, and may not be applicable to other time intervals as communities of birds change over time with forest succession following disturbance (Hobson and Schieck, 1999; Schieck and Song, 2006). Finally, abundance can be a misleading indicator of habitat quality (Van Horne, 1983), and our results do not inform how the treatments may have influenced breeding success, fitness, or population viability.

4.6. Management implications

We suspect that our observations of greater abundances of ground and shrub-nesting species and lower abundances of species associated with snags in the early years following postfire salvage logging are likely to be consistent throughout dry ponderosa pine and Douglas-fir forests in the interior west of North America. Responses of these species of birds to salvage logging in other forest types, fire severities and time since fire are uncertain.

Salvage logging influenced abundances of cup-nesting and cavity-nesting species. Morissette et al. (2002) similarly found that salvage logging influenced the numbers of cup-nesting birds in boreal forests of mixed-wood and jack pine after fire. The response of cup-nesting forest birds, especially sensitive and of conservation concern, should be considered in addition to the response of cavity-nesting species when implementing management prescriptions of salvage logging.

The black-backed woodpecker is a species of management concern and is designated as a “sensitive species” in Oregon (Oregon Natural Heritage Information Center, 2004) and Idaho (USDI, 2003), and a “species of concern” in Montana (Montana Natural Heritage Program and Montana Fish Wildlife and Parks, 2006). Our results support previous conclusions that the black-backed woodpecker is less abundant in salvaged forests than in unsalvaged forests (Caton, 1996; Saab and Dudley, 1998; Haggard and Gaines, 2001; Hutto and Gallo, 2006; Koivula and Schmiegelow, 2007; Saab et al., 2007); retention of areas of unsalvaged habitat may be of particular importance for this species.

Although we cannot determine whether greater abundances of some species in unsalvaged stands would result in greater nesting densities or increased population viability, our findings suggest that even relatively small patches of unsalvaged forest may provide benefits to species that respond negatively to salvage. Examining whether larger patches of unsalvaged forest similarly yield a positive or larger response would be important to determine the utility of small patches and tease apart potential edge effects in smaller patches.

Given the paucity of information on ecological responses to salvage logging, it is tempting to extrapolate information on responses of species to other management activities (e.g., thinning) in unburned forests to help predict likely responses of species to salvage harvest of burned forests. However, results from studies of bird abundance following thinning in unburned forests (Dellasala et al., 1996; Hagar et al., 1996, 2004; Artman, 2003; Hayes et al., 2003) do not closely parallel results from salvage logging of burned forests. While responses of the dark-eyed junco (Dellasala et al., 1996; Hagar et al., 1996, 2004; Artman, 2003; Hayes et al., 2003) to thinning and salvage harvest are qualitatively similar, our results for the western wood-pewee, hairy woodpecker and western tanager contrasted the results from studies of thinned forests (Hagar et al., 1996, 2004; Hayes et al., 2003). Differences in relative abundance of the brown creeper, red-breasted nuthatch, and American robin varied by study (Hagar et al., 1996, 2004; Artman, 2003; Hayes et al., 2003) and therefore are problematic to compare to our results. While responses for some species to the two management activities are similar, important differences exist that preclude easy extrapolation of results.

If the maintenance of native species of bird populations is a goal of forest management, than managers should leave some burned forest for birds that are postfire specialists, particularly forest that is otherwise suitable for salvage. Additionally, as influences of salvage logging are not limited to cavity-nesting species, the response of cup-nesting birds should also be considered when implementing salvage logging.

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Appendix A. Supplementary data

Supplementary data associated with this article can be found, in the online version, at doi:10.1016/j.foreco.2008.11.019.

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