Short-Term Effects of Thinning and Burning Restoration Treatments on Avian Community Composition, Density, and Nest Survival in the Eastern Cascades Dry Forests, Washington

William Gaines, Maryellen Haggard, James Begley, John Lehmkuhl, and Andrea Lyons

Abstract: We monitored the short-term (<3 years) response of land birds to restoration treatments (thin, burn, and thin-burn) in dry forests located on the eastern slope of the Cascade Range in Washington. Overall avian community composition did not change among the treatments. However, individual species responses varied with the chipping sparrow showing lower density in treatments, whereas hermit thrush, mountain chickadee, white-headed woodpecker, western bluebird, American crow, and common raven increased in treatment units. Daily survival rates of nesting guilds were similar in treated versus control stands; however, burn-only showed lower daily survival rates compared with other treatments. Additional research is needed to validate this result. Cavity-nesters (mountain chickadee, red-breasted nuthatch, and house wren) and foliage-nesters (chipping sparrow) used trees that were larger in diameter than available regardless of the treatment. Our results, in combination with other results from this study area, provide important implications for managers: (1) thin-burn treatments were effective at restoring habitat for several avian focal species; (2) spring burn treatments should be carefully designed to achieve desired restoration objectives; (3) large trees provide important habitat functions and are a key component for maintaining or restoring the viability of focal avian species; and (4) additional research is needed to better understand the effects of spring burning and the long-term effects of dry forest restoration treatments. FOR. SCI. 56(1):88–99.

Keywords: restoration treatments, dry forests, avian community composition, avian abundance and density, nest survival, focal avian species, mechanical, prescribed fire

FIRE EXCLUSION POLICIES in the 20th century created unnaturally high fuel profiles of dead and live fuels in many dry forests of the western U.S. and Canada (Allen et al. 2002, Brown et al. 2004). As a result, the area burned by wildland fire has significantly increased in the last 20 years primarily owing to accumulated fuels (Agee 1997, Fitzgerald 2002, Westerling et al. 2006). State and federal initiatives have been generated to deal with this issue (Western Governor's Association 2003, Healthy Forest Restoration Act of 2003). However, lack of information on the environmental effects of restoration treatments (e.g., thinning and burning) has complicated implementation. In 1999, a consortium of federal, nongovernment organization, and university scientists was funded by the US Department of Agriculture and US Department of Interior Joint Fire Science Program for a national network of 12 study sites where effects of restorative treatments could be studied using comparable methods. Our study in the Pacific Northwest is located on the Okanogan–Wenatchee National Forest in Washington State and is referred to as the Northeastern Cascades study area.

A growing number of studies have investigated the effects of restorative treatments in dry forests on avian species and communities (Germaine and Germaine 2002, Zebehazy et al. 2004, Wightman and Germaine 2006, Gaines et al. 2007, Greenberg et al. 2007). Generally, these treatments have little effect on overall species abundance, richness, or evenness (Zebehazy et al. 2004, Gaines et al. 2007, Greenberg et al. 2007). However, responses of individual avian species vary. For example, species associated with more open forest conditions, as occurred in much of the dry forest before fire exclusion (Harrod et al. 1999, Hessburg et al. 1999, Agee 2003, Wright and Agee 2004), such as the western bluebird (*Sialia mexicana*) and white-headed woodpecker (*Picoides albolarvatus*), respond favorably to these treatments (Wightman and Germaine 2006, Gaines et al. 2007). Conversely, species associated with more closed forest conditions, such as the red-breasted nuthatch (*Sitta canadensis*) may occur at lower densities in treated stands (Gaines et al. 2007). Reasons that have been cited for the positive response of species to treatments include increased herbaceous and bare ground cover resulting in more diverse

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invertebrate assemblages and greater food abundance (Wightman and Germaine 2006, Gaines et al. 2007, Greenberg et al. 2007). The effects of these kinds of restoration treatments on wildlife need to be understood because it is imperative that forest restoration not focus solely on forest structural attributes without consideration of impacts to other forest ecosystem processes and functions (Germaine and Germaine 2002, Lehmkuhl et al. 2007b, Russell et al. 2009).

Our objectives were to describe the effects of restoration treatments (prescribed fire, thinning, and thinning followed by prescribed fire) on avian community structure, density, and nesting survival. Gaines et al. (2007) studied the effects of ponderosa pine restoration treatments on the avian community in a nearby study area. Our study differs from that of Gaines et al. and builds on their work by reporting avian responses to dry forest restoration treatments: in a wider range of dry forest types, from ponderosa pine plant associations to dry grand fir plant associations (Harrod et al. 2007); for burn-only and thin-only treatments in addition to a combined thinning and burning treatment; and in terms of nest survival, an important factor in understanding the effects of these treatments.

In this study we addressed two specific hypotheses that we developed from previous research results from similar environments and treatments. First, we expected that species responses (density, abundance, and nest survival) to the treatments would be variable with a general pattern that species associated with open-canopy forests (such as western bluebird and white-headed woodpecker) would respond positively to dry forest restoration treatments, whereas those associated with closed-canopy forests (such as red-breasted nuthatch and brown creeper [*Certhia americana*]) would not. Second, we expected that large trees, found to be important for nesting (Bull et al. 1997, Bevis and Martin 2002, Bunnell et al. 2002, Dickson et al. 2004) and foraging (Lyons et al. 2008), would be important in our study area, irrespective of the treatment applied.

Methods

Study Area

The Northeastern Cascades study area (Figure 1) is located in the eastern Cascade Range of central Washington state within the Okanogan-Wenatchee National Forest. Climate is continental because of the rain shadow produced by the Cascade Range. Annual precipitation averages 22 cm, falling mostly as snow between November and April. Historically, forests within the study area were dominated by ponderosa pine (*Pinus ponderosa*) and Douglas-fir (*Pseudotsuga menziesii*) with grand fir (*Abies grandis*) and western larch (*Larix occidentalis*) occurring at higher elevations (Agee 1994, Harrod et al. 1999). Potential vegetation consists of dry to mesic Douglas-fir plant association groups (Lillybridge et al. 1995), and we constrained the location of sample units to within these forested plant associations. Additional details on the composition of the forested vegetation types can be found in Harrod et al. (2007) and Dodson et al. (2008).

Fire exclusion and logging have significantly altered

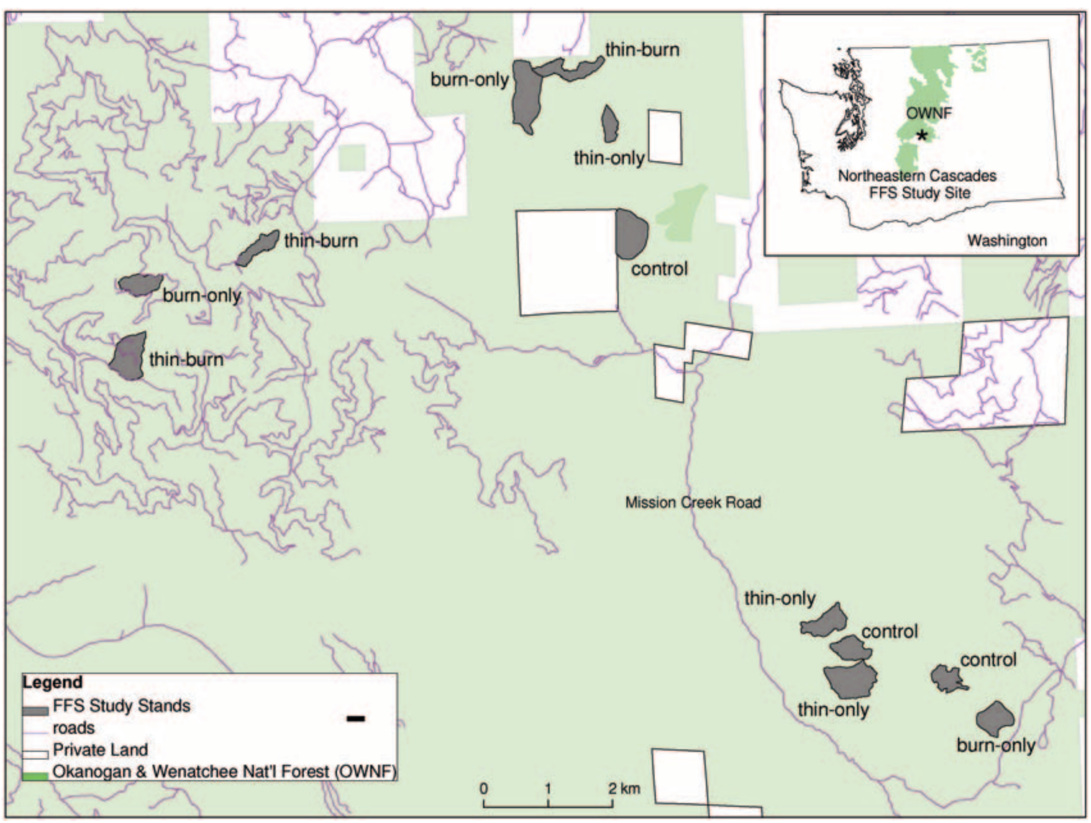

Figure 1. Northeastern Cascades Fire and Fire Surrogate study site showing the study stands and treatment assignments.

these forests (Harrod et al. 1999, Hessburg and Agee 2003, Wright and Agee 2004). Past logging practices in the project area removed many fire-tolerant large (>50 cm diameter) ponderosa pine trees. The result is that much of the study area is composed of small (<30 cm diameter) trees at higher densities (≥10× more trees/ha) that are more prone to high-severity fires and epidemic insect outbreaks (Harrod et al. 1999). Elevations within the project area range from 600 to 1,200 m, and annual precipitation averages approximately 50 cm.

Study Design

General methods for all National Fire and Fire Surrogate (FFS) study sites were developed as part of the national meta-study (McIver et al. 2008). Approximately 30 candidate units were identified in 1999 in the Mission Creek watershed and another smaller watershed immediately adjacent to the west. We constrained the search to eliminate units that were <10 ha in size with north aspects, steep slopes (>40%), >10% rock or nonforest vegetation cover, and known plant or animal species of concern. In the Northeastern Cascades study area, 12 of a possible 30 potential units, each about 10 ha in size, were chosen as experimental units and were randomly assigned as treatments and controls. Treatments included thinning from below (thin-only), prescribed fire alone (burn-only), commercial thinning from below followed by prescribed fire (thin-burn), and untreated control. Treatments were assigned to the 12 study units, producing three replicates of each.

Unit Treatment

The specific treatment objectives were to restore low-density dry forest stand structure, reduce ladder fuels and surface fuels, reduce the risk of extensive bark beetle attack, and reduce the risk of high severity fires. Thinning was designed to reduce basal area to 10–14 m^2 ha^{-1} in a nonuniform pattern to mimic natural stand patterns and increase resistance to bark beetle attack (Harrod et al. 1999). Thinning was completed in spring 2003. Trees were removed, concentrating on smaller commercial tree sizes, until the desired basal area was reached. Yarding was done by helicopter because of steep slopes and limited road access, resulting in branches and tops being left on site. Thinned units were slashed (mechanically cut by hand) after harvest to fall smaller, unmerchantable stems. Burning was completed in the spring of 2004. Ignition of burn units was by hand and helicopter, resulting in flame lengths ranging from 0.2 to 1.0 m. Because of early green-up in 2004, burn coverage was spotty, ranging from 23 to 51% of the treatment unit (Agee and Lolley 2007). Early green-up and high fuel moisture also caused two of the six scheduled fires to be postponed (completed in 2006). As a result, the burn-only and thin-burn treatments had only two units each, and the remaining unburned units were added to the control and thin-only treatments, giving them four units each. The pretreatment basal area ranged from 22.7 to 42.7 m^2/ha. The thin-burn treatments reduced basal area by 39–46%, thin-only treatments reduced basal area by 34–69%, and the burn-only treatment did not measurably reduce basal area.

Avian Species Abundance

Point counts are a relatively standardized method for estimating the relative abundance and diversity of avian species (Ralph et al. 1993, Buckland et al. 2001). Point counts were conducted after treatments during 2004 and 2005 from early May to late June at 4–6 points/stand for a total of 4 visits to each of the 12 study stands. Point count centers were located at least 200 m apart and at least 100 m from the stand edge. Each point count began within ½ hour of the official sunrise and was completed no later than 10:00 am PST. After arriving at the point, the observer waited at least 2 minutes and then identified and counted birds for 10 minutes. Birds were identified by song, call, or visual detections. Detections of birds were recorded at 10-m increments out to 80 m using horizontal distance. Flyovers were documented but were not used in the analyses. Standard four-letter codes for each bird species were recorded.

A complete count of all stands was finished before the second visit was made to any stand. Logistical constraints prevented a completely random sequence of sampling. Some stands were grouped together (Crow 1, Crow 3, and Crow 6) and sampled on a single day to improve sampling efficiency. All point-count stations were geo-referenced using global positioning systems.

Nest Survival and Substrate

We used standardized methods to assess avian productivity and survival (Martin and Geupel 1993, Ralph et al. 1993). In the Northeastern Cascades study area nest searching and monitoring were conducted from early May until mid-July. Nest searches were conducted in two replicates of each treatment (including controls), and nests were monitored until the fate (fledging or failure) was determined. Stands were thoroughly searched for nests following routes that traversed all parts of the stands. Once a nest was found, flagging was used (10–15 m away) to indicate the species and nest number. Detailed instructions with a drawing were made so that the nest could be relocated for subsequent monitoring.

Nests were monitored from a distance, whenever possible, and all efforts were made to minimize disturbance. Nests were checked every 2–4 days, keeping careful track of the stage of each nest. Species-specific literature on clutch sizes, incubation, and nestling periods was used to estimate when incubation, hatching, or fledging was likely to occur so that more frequent visits were made during these times. Dead-end paths were avoided by entering along one path and exiting along another when nests were checked so that predators had difficulty determining the exact nest location. Active nests were not visited if predators were observed nearby.

Tree size (dbh) and tree species were measured for each nest tree and compared with those for the available trees and snags in each study stand. This was done for cavity-nesters and foliage-nesters with sufficient sample sizes: mountain chickadee (*Parus gambeli*) (cavity-nester), red-breasted nuthatch (cavity-nester), house wren (*Troglodytes aedon*) (cavity-nester), and chipping sparrow (*Spizella passerine*) (foliage nester). The availability of tree and snag habitat was assessed from data in Harrod et al. (2007).

Statistical Analyses

We examined the compositional similarity of the avian communities among the treatments using a nonparametric multivariate analysis in PC-ORD4 software (McCune and Mefford 1999). We chose this method over univariate measures because it summarizes all of the information on the bird community simultaneously, allowing for an assessment of the community level response to the treatments (Hanowski et al. 2003). For the community level analyses, we calculated an index of relative species abundance (detections/point/visit/year) for each treatment unit across both study years (2004 and 2005) using detections recorded ≤50 m from point centers for species with >1 detection. Examination of the detection probabilities at 10-m increments out to 100 m (using Program DISTANCE) for several species among treatments indicated that they were nearly equal out to 60–70 m. Based on this finding, we assumed that the 50-m sampling area is sufficiently small to overcome potential detectability issues related to variation among bird species and treatments (Sallabanks et al. 2006).

We used multiresponse permutation procedures to test the hypothesis of no difference in avian species composition between the treatment types (Zimmerman et al. 1985, Biondini et al. 1988). This produced an A statistic that measured the grouping "effect size," or distinctiveness of groups, on a scale of 0–1. Values of $A > 0.3$ are considered fairly high. Monte Carlo permutations were used to calculate probabilities for differences among the treatments. We compared those probabilities with Bonferroni-adjusted P values obtained by dividing the experiment-wise error rate $P \leq 0.05$ by three comparisons.

We used indicator species analysis (ISA), in PC-ORD4 (McCune and Mefford 1999), to identify characteristic species found mostly in a single type (treatment) and present in the majority of the sites (replicates) belonging to that type (Dufrene and Legendre 1997). This analysis approach provides important information for managers about which species are most likely to respond to treatments, potentially making them good candidates for monitoring. In addition, we chose ISA because it combined information on both species relative abundance and constancy to estimate indicator values for each species in each group. The maximum indicator value for a species within the treatment was tested for statistical significance against random expectation calculated by Monte Carlo permutation.

To evaluate individual species responses to the restoration treatments we used Program DISTANCE to estimate bird density and assess detection probabilities of several species (Buckland et al. 2001). Data were pooled across sample years (2004 and 2005) to increase sample sizes and the number of species for which we could calculate density estimates. We followed guidelines described in Buckland et al. (2001) to determine adequate sample sizes for estimating density. Generally, this required >30 detections per treatment for each species. We calculated mean density and 95% confidence intervals (CI) using density estimates from the replicates of each treatment. We analyzed for treatment effects using analysis of variance and performed multiple comparisons using Tukey's honestly significant difference test (SAS Enterprise 3.0) (Nur et al. 1999).

We calculated daily survival rates (Mayfield 1961, 1975, Nur et al. 1999) for all species within nesting guilds. We assumed constant survival over the nesting period (Donovan et al. 1995, Annand and Thompson 1997). We calculated 95% CI for each survival estimate and used these to assess for differences in survival of nesting guilds among treatments (Nur et al. 1999).

We used χ^2 tests (with Yates correction when $v = 1$) (Zar 1996) to examine differences in used versus available nest tree species. Nest tree size was compared with available tree sizes using analysis of variance (SAS Enterprise 3.0). All significance tests used $P = 0.05$.

Results

Avian Community Composition

We recorded a total of 5,748 detections of 85 avian species. We had a sufficient number of detections to estimate density for seven species, of which only two showed any differences between treated and control stands (Table 1). Abundance indices were derived for 49 species (Table 2), which also were used in the indicator species analyses (Table 3). The most common species detected were the chipping sparrow, dark-eyed junco (*Junco hyemalis*),

Table 1. Density estimates of avian species by treatment within the Northeastern Cascades Fire and Fire Surrogate study

Species	Control		Burn-only		Thin-only		Thin-burn	
	Density	95% CI	Density	95% CI	Density	95% CI	Density	95% CI
	(No./ha)							
Chipping sparrow	5.97	1.98–18.00	0.46[a]	0.36–0.59	2.64	0.79–8.81	3.14	1.50–6.57
Dark-eyed junco	6.20	3.75–10.26	9.35	4.94–17.68	6.14	2.72–13.84	8.57	3.42–21.46
Mountain chickadee	0.39[b]	0.32–0.47	0.57	0.28–1.16	0.45	0.20–1.03	1.15[b]	0.53–2.49
Nashville warbler	0.31	0.12–0.85	0.4	0.27–0.59	0.47	0.20–1.11	0.53	0.22–1.29
Red-breasted nuthatch	0.49	0.39–0.61	0.62	0.45–0.86	0.52	0.30–0.89	0.53	0.26–1.07
Western tanager	0.56	0.45–0.70	0.62	0.49–0.78	0.73	0.56–0.95	0.66	0.37–1.19
Yellow-rumped warbler	0.88	0.70–1.12	1.37	0.43–4.35	0.97	0.58–1.64	0.97	0.54–1.73

[a] Chipping sparrow: significantly ($P < 0.05$) lower density in burn-only compared with all other treatments.
[b] Mountain chickadee: significantly ($P < 0.05$) lower density in control versus thin-burn treatments.

Table 2. Relative abundance and 95% CI of avian species (detections/point/visit/year) by treatment in the Northeastern Cascades Fire and Fire Surrogate study, 2004 and 2005

	Control		Thin-only		Thin-burn		Burn-only	
Species	Mean	95% CI	Mean	95% CI	Mean	95% CI	Mean	95% CI
Low understory ground insectivore								
Chipping sparrow	1.24	0.50	1.06	0.79	1.02	0.57	0.52	0.03
Dark-eyed junco	1.28	0.09	1.15	0.33	1.35	0.78	1.59	0.06
Townsend's solitaire	0.27	0.09	0.41	0.21	0.53	0.16	0.54	0.45
American robin	0.57	0.23	0.46	0.39	1.01	0.24	0.86	0.46
Brown-headed cowbird	0.05	0.05	0.10	0.14	0.11	0.14	0.05	0.04
House wren	0.09	0.10	0.36	0.38	0.13	0.24	0.07	0.01
Nashville warbler	0.27	0.23	0.42	0.35	0.56	0.48	0.30	0.09
Black-headed grosbeak	0.09	0.11	0.06	0.05	0.17	0.11	0.05	0.09
Spotted towhee	0.39	0.05	0.31	0.23	0.18	0.11	0.33	0.47
Hermit thrush	0.08	0.09	0.06	0.04	0.08	0.09	**0.43**	**0.50**
MacGillivray's warbler	0.14	0.12	0.23	0.25	0.23	0.23	0.56	0.37
Purple finch	0.01	0.02	0.05	0.07	0.00	0.00	0.04	0.04
Swainson thrush	0.19	0.18	0.12	0.14	0.18	0.11	0.21	0.32
Tree foliage insectivore								
Yellow-rumped warbler	1.14	0.25	1.14	0.55	1.04	0.54	1.10	1.27
Western tanager	1.05	0.17	1.08	0.24	1.00	0.56	1.18	0.18
Mountain chickadee	0.60	0.07	0.44	0.16	0.97	0.46	0.80	0.45
Townsend's warbler	0.19	0.16	0.30	0.49	0.28	0.34	0.21	0.26
Cassin's vireo	0.59	0.28	0.43	0.09	0.40	0.19	0.72	0.24
Golden-crowned kinglet	0.05	0.08	0.08	0.06	0.03	0.06	0.15	
Ruby-crowned kinglet	0.01	0.01	0.09	0.17	0.10	0.17	0.08	0.02
Wilson's warbler	0.04	0.06	0.02	0.03	0.00	0.00	0.00	0.00
Lazuli bunting	0.01	0.02	0.00	0.00	0.01	0.02	0.02	0.03
Warbling vireo	0.03	0.06	0.11	0.16	0.02	0.02	0.13	0.24
Cedar waxwing	0.01	0.02	0.04	0.06	0.00	0.00	0.00	0.00
Bark insectivore								
Red-breasted nuthatch	1.04	0.15	0.92	0.53	1.00	0.71	1.07	0.31
Hairy woodpecker	0.06	0.04	0.09	0.09	0.20	0.18	0.08	0.02
White-breasted nuthatch	0.03	0.04	0.09	0.11	0.07	0.11	0.01	0.02
Pygmy nuthatch	0.07	0.06	0.13	0.13	0.09	0.06	0.03	0.01
White-headed woodpecker	0.00	0.00	0.01	0.02	**0.11**	**0.07**	0.00	0.00
Brown creeper	0.05	0.02	0.03	0.02	0.07	0.02	0.04	0.04
Northern flicker	0.01	0.02	0.04	0.04	0.07	0.09	0.01	0.02
Pileated woodpecker	0.01	0.01	0.00	0.00	0.03	0.04	0.00	0.00
Aerial insectivore								
Western bluebird	0.00	0.00	0.11	0.13	0.22	0.43	0.00	0.00
Pacific-slope flycatcher	0.00	0.00	0.01	0.02	0.00	0.00	0.03	0.05
Western wood-pewee	0.09	0.10	0.25	0.36	0.09	0.12	0.04	0.07
Dusky flycatcher	0.38	0.39	0.44	0.30	0.25	0.23	0.41	0.36
Hammond's flycatcher	0.27	0.31	0.16	0.19	0.06	0.06	0.10	0.05
Dusky/Hammond's flycatcher	0.38	0.26	0.21	0.15	0.19	0.16	0.19	0.12
Olive-sided flycatcher	0.01	0.02	0.00	0.00	**0.04**	**0.02**	0.02	0.03
Gray flycatcher	0.07	0.14	0.12	0.13	0.07	0.14	0.00	0.00
Tree seedeater								
Cassin's finch	0.61	0.27	0.73	0.35	0.93	0.36	0.54	0.47
Pine siskin	0.34	0.17	0.36	0.12	0.41	0.11	0.26	0.22
Red crossbill	0.11	0.09	0.11	0.10	0.23	0.33	0.03	0.01
Omnivore-scavenger								
Gray jay	0.00	0.00	0.02	0.03	0.04	0.04	0.00	0.00
Steller's jay	0.02	0.02	0.00	0.00	0.02	0.04	0.02	0.03
American crow	0.01	0.01	0.01	0.01	**0.08**	**0.04**	0.03	0.05
Common raven	0.03	0.04	0.03	0.04	**0.16**	**0.09**	0.03	0.05
Nectivore								
Calliope hummingbird	0.00	0.00	0.05	0.05	0.02	0.02	0.03	0.01
Rufus hummingbird	0.03	0.06	0.01	0.02	0.01	0.02	0.02	0.03

Species listed include those with >1 detection and with detection distances truncated to 50 m. Significant differences ($P < 0.05$) are shown in bold.

yellow-rumped warbler (*Dendroica coronata*), western tanager (*Piranga ludoviciana*), and red-breasted nuthatch. Neotropical and migratory bird species comprised 51% of the total number of species (>1 detection), 52% of individual birds detected in the control stands, and 49% of the detections in treated stands.

Avian species assemblages did not differ among the treatments (multiresponse permutation procedure A = 0.026, $P = 0.589$). However, there were some differences in individual species densities and detections among the control and treatments. The chipping sparrow density was 12 times higher in the control units than in the burn-only units

Table 3. **Indicator species values for control and treatments in the Northeastern Cascades Fire and Fire Surrogate study**

Species	Indicator species values				P[a]
	Control	Thin	Thin-burn	Burn	
Low understory/ground insectivore					
Chipping sparrow	28	27	13	32	0.605
Dark-eyed junco	21	25	30	24	0.456
Townsend's solitaire	23	30	31	16	0.618
American robin	16	35	30	20	0.275
Brown-headed cowbird	32	35	16	16	0.920
House wren	56	20	11	14	0.204
Nashville warbler	27	36	19	18	0.481
Black-headed grosbeak	16	46	13	25	0.203
Spotted towhee	25	15	28	32	0.692
Hermit thrush	10	13	**65**	12	**0.019**
MacGillivray's warbler	20	20	48	12	0.112
Purple finch	47	0	43	10	0.319
Swainson thrush	16	25	30	28	0.881
Winter wren	100	0	0	0	1.000
Varied thrush	0	100	0	0	0.650
Tree foliage insectivore					
Yellow-rumped warbler	26	24	25	26	0.996
Western tanager	25	23	27	24	0.788
Mountain chickadee	16	35	29	21	0.245
Townsend's warbler	30	29	21	19	0.936
Cassin's vireo	20	19	34	28	0.145
Golden-crowned kinglet	27	10	48	16	0.171
Ruby-crowned kinglet	32	36	29	2	0.752
Veery	0	0	0	100	0.676
Wilson's warbler	29	0	0	71	0.190
Lazuli bunting	0	29	43	29	0.825
Warbling vireo	39	7	43	11	0.923
Mourning dove	69	0	0	31	1.000
Cedar waxwing	80	0	0	20	0.282
Bark insectivore					
Red-breasted nuthatch	23	25	26	26	0.990
Hairy woodpecker	21	46	20	13	0.268
White-breasted nuthatch	44	35	6	15	0.865
Pygmy nuthatch	40	29	9	22	0.878
White-headed woodpecker	10	**90**	0	0	**0.004**
Brown creeper	15	37	22	26	0.172
Northern flicker	28	55	9	8	0.443
Red-naped sapsucker	53	0	0	47	1.000
Black-backed woodpecker	0	100	0	0	0.650
Pileated woodpecker	0	82	0	18	0.130
Aerial insectivore					
Western bluebird	34	66	0	0	0.636
Pacific-slope flycatcher	29	0	71	0	0.251
Western wood-pewee	53	20	8	19	0.960
Dusky flycatcher	30	17	28	25	0.960
Hammond's flycatcher	27	11	17	46	0.354
Olive-sided flycatcher	0	62	23	15	0.113
Violet-green swallow	65	0	35	0	1.000
Gray flycatcher	45	28	0	28	0.647
Tree seedeater					
Cassin's finch	26	33	19	22	0.314
Pine siskin	27	30	19	25	0.577
Red crossbill	23	48	6	23	0.510
Evening grosbeak	0	0	0	100	0.684
Omnivore-scavenger					
Gray jay	27	73	0	0	0.189
Steller's jay	0	39	29	32	0.808
American crow	4	**69**	21	6	**0.030**
Common raven	13	**64**	10	13	**0.013**
Nectivore					
Calliope hummingbird	49	22	29	0	0.364
Rufous hummingbird	19	15	22	44	1.000

[a] P values are based on the maximum indicator values for the species among the treatments. Bold numbers show values for indicator species ($P < 0.05$).

(Table 1). The hermit thrush (*Catharus guttatus*) had a higher indicator species value ($P = 0.019$) in the thin-burn treatments and was 5 times greater in abundance ($F = 3.5$, $P = 0.07$) in the burn-only treatment compared with values for the control units (Tables 2 and 3). Mountain chickadee had a density that was 3 times higher in the thin-burn treatment compared with the density in the controls (Table 1). The white-headed woodpecker ($F = 7.83$, $P = 0.009$; ISA $P = 0.004$) and olive-sided flycatcher (*Contopus cooperi*) ($F = 4.61$, $P = 0.037$) had higher abundance and indicator species value for thinning treatments (both thin-only and thin-burn) (Tables 2 and 3). The western bluebird was only detected in the thin-only and thin-burn treatments (Table 2). Finally, both American crows (*Corvus brachyrhynchos*) ($F = 7.52$, $P = 0.01$; ISA $P = 0.03$) and common ravens (*Corvus corax*) ($F = 4.22$, $P = 0.04$; ISA $P = 0.01$) had greater abundance (8 and 5 times higher, respectively, compared with control stands) and higher indicator species values for treated stands (burn-only, thin-only, and thin-burn) (Tables 2 and 3).

Nest Survival and Substrate

We located and monitored a total of 175 nests representing 24 avian species during 2 years of nest searching. Most of the nests were located in the thin-only treatment (77 nests, 44%), and the remaining nests were fairly equally distributed among the burn-only (37 nests, 21%), thin-burn (32 nests, 18%), and control (29 nests, 17%) treatments. Three nesting guilds were represented in the sample of nests that were monitored: 76 (43%) cavity nests, 68 (39%) foliage nests, and 31 (18%) ground nests. The most common ground-nester was the dark-eyed junco (23 nests), the most common foliage-nester was the chipping sparrow (21 nests), and the most common cavity-nesters were the red-breasted nuthatch (15 nests) and mountain chickadee (14 nests).

We did not have sufficient sample sizes for any species to analyze for treatment effects on survival. Thus, we compared survival rates for each nesting guild among the treatments (Table 4). Daily survival rates were higher for cavity-nesting species than for foliage- and ground-nesters, regardless of treatment (Table 4). Survival rates varied little among the treatments with the exception that survival was lower for ground-nesters in the burn-only treatments (Table 4). We detected only one possible incident of nest parasitism by brown-headed cowbirds (*Molothrus ater*) while implementing our nest monitoring protocol.

Cavity-nesters (mountain chickadee [$F = 14.06$, $P = 0.0022$], red-breasted nuthatch [$F = 10.36$, $P = 0.0074$], and house wren [$F = 10.71$, $P = 0.0113$]) and foliage-nesters (chipping sparrow [$F = 16.94$, $P = 0.0012$]) used nest trees larger in diameter (dbh) than available in the study units, regardless of the treatment (Table 5). The mountain chickadee used ponderosa pine trees greater than available for nests in all treatments except the thin-burn treatment where other tree species (western larch and grand fir) were used (Table 5). Red-breasted nuthatch used tree species equal to those available in control stands, Douglas-fir greater than available in burn-only and thin-burn stands, and ponderosa pine greater than available in thin-only stands (Table 5). House wren nests were only located within the thin-only treatment, and they used ponderosa pine trees greater than available for nesting (Table 5). The chipping sparrow used ponderosa pine greater than available in control and thin-burn stands, Douglas-fir greater than available in burn-only stands, and tree species equal to availability in the thin-only treatment (Table 5).

Table 4. Daily nest survival rates (DSR) by nesting guild for avian species within the Northeastern Cascades Fire and Fire Surrogate study

Functional group	*n* (nests)	No. of observation days	DSR ± SE
Ground-nesting species			
Control	6	48	0.958 ± 0.029
Burn	9	71	0.915 ± 0.033
Thin	10	130	0.977 ± 0.013
Thin-burn	6	99	0.980 ± 0.014
Foliage-nesting species			
Control	15	184	0.962 ± 0.014
Burn	14	160	0.969 ± 0.014
Thin	27	437	0.977 ± 0.007
Thin-burn	12	209	0.981 ± 0.009
Cavity-nesting species			
Control	8	123.5	0.992 ± 0.008
Burn	14	234	0.987 ± 0.007
Thin	40	588.5	0.986 ± 0.005
Thin-Burn	14	209	1.0 ± 0.0

Discussion

Our results, similar to those reported by Hurteau et al. (2008), did not indicate any significant differences in overall species assemblages among the four treatments, although we did detect different responses by some individual species. Gaines et al. (2009) did not detect any differences in the pretreatment bird community among the stands. No difference in avian communities among treatments varies from those reported in Gaines et al. (2007), in which avian communities in ponderosa pine stands treated with combined thinning and under-burning differed from those in control stands. One possible explanation for these differing results was that the treatments in Gaines et al. (2007) all included thinning followed by burning. These treatments generally resulted in greater changes to forest structure and understory composition than either thinning or burning alone (Agee and Lolley 2007, Harrod et al. 2007, Dodson et al. 2008), which were included as treatments in our study.

The chipping sparrow was identified as a focal species for ponderosa pine forests in the East-slope Cascade Mountains conservation strategy (Altman 2000). Their populations have been reported as declining in the Cascade Mountains physiographic province (Sauer et al. 1999). Sallabanks et al. (2006) reported that open-canopy stands with understories of grass and shrubs were used most frequently by the chipping sparrow in the Blue Mountains of Oregon. Whereas Gaines et al. (2007) found that chipping sparrow densities increased in treated stands, we found few differences in our treatments. The chipping sparrow occurred at a lower density in the burn-only treatments; however, the

Table 5. Nest tree size and species used by cavity-nesters and foliage-nesters compared to the available tree sizes and species composition within each treatment, Northeastern Cascades Fire and Fire Surrogate study, 2004–2005

Species	Control	Burn	Thin	Thin-burn
Mountain chickadee (n = 14): cavity-nester				
Nest tree size (cm dbh ± SE)	65.6 ± 8.9	56.3 ± 8.3	49.1 ± 11.8	**34.0 ± 18.5**
Nest tree species (% of total nests)				
Ponderosa pine	**80**	**100**	**100**	**0**
Douglas-fir	**20**	**0**	**0**	**50**
Other[a]	**0**	**0**	**0**	**50**
Red-breasted nuthatch (n = 15): cavity-nester				
Nest tree size (cm dbh ± SE)	29.4 ± 9.2	44.9 ± 3.5	46.5	**47.7 ± 9.5**
Nest tree species (% of total nests)				
Ponderosa pine	33	**50**	**100**	**29**
Douglas-fir	67	**50**	**0**	**71**
Other[a]	0	**0**	**0**	**0**
House wren (n = 19): cavity-nester				
Nest tree size (cm dbh ± SE)	0	0	38.5 ± 2.5	**0**
Nest tree species (% of total nests)				
Ponderosa pine	0	0	**100**	**0**
Douglas-fir	0	0	**0**	**0**
Other[a]	0	0	**0**	**0**
Chipping sparrow (n = 14): foliage-nester				
Nest tree size (cm dbh ± SE)	45.7 ± 9.2	44.2 ± 10.1	62.2 ± 4.9	**36.2 ± 2.7**
Nest tree species (% of total nests)				
Ponderosa pine	**67**	**75**	60	**50**
Douglas-fir	**33**	**25**	40	**50**
Other[a]	**0**	**0**	0	**0**
Available				
Tree size (cm dbh ± SE)	25.7 ± 1.3)	24.2 ± 1.9	23.8 ± 1.7	**24.8 ± 2.7**
Tree species (% of total nests)				
Ponderosa pine	28	87	58	**27**
Douglas-fir	72	13	42	**61**
Other[a]	**0**	**0**	**0**	**12**

Numbers in bold show significant differences (P = 0.05) between used and available habitats.
[a] Includes grand fir and western larch.

burn treatments occurred in the spring and were very spotty, resulting in little effect on forest structure (Agee and Lolley 2007, Harrod et al. 2007). Perhaps spring burning could have influenced chipping sparrow nesting as they are foliage nesters (Baicich and Harrison 1997), and the burn-only treatment did reduce lower limbs, raising canopy height (Harrod et al. 2007). Regardless of the treatment, the chipping sparrow selected large trees for nesting habitat but used both ponderosa pine and Douglas-fir species.

Hermit thrush is a focal species in the East-slope Cascade Mountains conservation strategy for multilayered, structurally diverse mixed-conifer forests (Altman 2000). Gaines et al. (2009) predicted that the hermit thrush would be negatively affected by the restoration treatments. However, we found that hermit thrush had high indicator species values in the thin-burn treatments and greater abundance in the burn-only treatment. It appears that the relatively light treatments implemented in this study had little to positive effects on the hermit thrush. The reported mixed responses for hermit thrush to thinning treatments may make them less suitable to serve as focal species for multilayered, structurally diverse mixed-conifer forests than was thought previously (Altman 2000, Lehmkuhl et al. 2007a).

Other studies have reported mixed responses of mountain chickadees to prescribed fire (Horton and Mannan 1988, Finch et al. 1997) and light overstory removal silvicultural treatments (Anderson and Crompton 2002, Finch et al. 1997). Mountain chickadees were associated with high canopy closure in northeast Oregon (Mannan and Meslow 1984, Sallabanks et al. 2002) and in west-central Idaho (Medin 1985). Gaines et al. (2007) and Hurteau et al. (2008) reported that restoration treatments in ponderosa pine forests resulted in lower densities of mountain chickadee. They cited the loss of snag habitat that may have been suitable for nesting as a potential reason for the lower densities in treated stands. Harrod et al. (2007) reported significant increases in the densities of snags within the thin-burn treatment at our study site, which may be the reason that mountain chickadees occurred in higher densities in this treatment and explain why this result varied from those reported in Gaines et al. (2007). Mountain chickadees selected ponderosa pine over Douglas-fir for nesting but also used western larch and grand fir when available, independent of treatments. Large trees were selected as nesting habitat in all treatments and were also important as a foraging habitat for chickadees (Lyons et al. 2008).

The white-headed woodpecker is a focal species for ponderosa pine forests in the East-slope Cascade Mountains conservation strategy (Altman 2000) for the Pacific Northwest region of the Forest Service (US Forest Service 2006) and is a Sensitive Species in Oregon and Washington (US Forest Service 2008). Broad-scale habitat declines have been reported for this species, especially in the North Cascades (Wisdom et al. 2000). Several researchers have suggested that active management, such as thinning and prescribed burning, be used to restore their habitat (Marshall

1997, Gaines 2000, Wisdom et al. 2000). Gaines et al. (2007) reported that restoration treatments applied to ponderosa pine forests enhanced habitat conditions for the white-headed woodpecker because they were only detected in treated stands. A similar result was found in this study in which white-headed woodpeckers were only detected in thin-only and thin-burn treatments and had higher indicator species values within thinned stands. The white-headed woodpecker was detected in very low abundance in pretreatment stand conditions (Gaines et al. 2009). These results, along with those of Gaines et al. (2007), provide support for the use of restoration treatments to enhance habitat conditions for white-headed woodpeckers within ponderosa pine and mixed-conifer dry forests. However, the lack of detection within the burn-only treatment suggests that the fire was not intense enough to reduce the density of small trees (Agee and Lolley 2006, Harrod et al. 2007) and provide habitat conditions that were open enough to encourage use by white-headed woodpeckers.

The olive-sided flycatcher is a focal species in the East-slope Cascade Mountains conservation strategy for edges and openings within mixed-conifer forests (Altman 2000). Wisdom et al. (2000) reported a moderate increase (29%) in source habitat for the olive-sided flycatcher in the North Cascades. Previous studies have reported positive responses of this species to timber harvest treatments (Medin and Booth 1989, Evans and Finch 1994), consistent with our findings of positive effects from restoration treatments. These positive responses are probably a result of more open forest canopies providing more foraging opportunities for this aerial insectivore.

The western bluebird was used as a focal species in the interior Columbia Basin Assessment (Wisdom et al. 2000) and for the US Forest Service Pacific Northwest Region (US Forest Service 2006). Declines have been noted for this species in many parts of its range (Ehrlich et al. 1988), and strong declines have been reported in the availability of source habitat in the North Cascades (Wisdom et al. 2000). Fire exclusion leading to a decrease in open-forest habitat has been cited as one reason for this decline (Guinan et al. 2000, Wisdom et al. 2000). Germaine and Germaine (2002) found that in restoration treatments in ponderosa pine forests of Arizona, western bluebird had a higher probability of successfully fledging young but were at greater risk of parasitic infections that reduce postfledging survival. Wightman and Germaine (2006) reported that habitat variables associated with higher nest success included increased herbaceous and bare ground cover and reduced ponderosa pine densities. Gaines et al. (2007), as in this study, only detected western bluebirds within treated stands, where more open-forest conditions were created. They were not detected within any of the pretreatment surveys (Gaines et al. 2009). Similar to the findings for white-headed woodpecker, the burn-only treatment did not create conditions that were open enough for western bluebirds. Restorative treatments that reduce canopy closure may provide bluebirds with open perches from which to hunt and thereby improve the availability of invertebrates as a food source (Wightman and Germaine 2006, Hurteau et al. 2008).

American crows, common ravens, and brown-headed cowbirds can act as nest predators or parasites and are often found to benefit from human-induced disturbances (Patton 1994, Hartley and Hunter 1998, Tewksbury et al. 1999). We found no evidence that the numbers of brown-headed cowbirds were influenced by treatments, similar to results reported in Gaines et al. (2007). There were no pretreatment differences in cowbird numbers among the stands (Gaines et al. 2009). We did find that American crows and common ravens had strong indicator values for treated stands (Table 3). However, we found little evidence in any stands of nest predation or parasitism.

Neotropical and migratory birds comprised a sizeable portion of the avian community in this study: 52% in control stands and 49% in treated stands. These percentages compare with 47% of the avian community in ponderosa pine forests (Gaines et al. 2007), 33% in mature Douglas-fir forests and 45% in young Douglas-fir forests of the southern Washington Cascade Range (Manuwal 1991), and 70% in the oak woodlands of south-central Washington (Manuwal 2003). Our results are similar to those of Gaines et al. (2007) in that we found restoration treatments have either neutral or positive effects on the abundance of many neotropical and migratory species.

As others have reported, we consistently found that cavity-nesters were more successful in completing nesting attempts compared with ground- or foliage-nesters (Zebehazy et al. 2004, Gaines et al. 2009). The only treatment that showed reduced daily survival rates in this study was the burn-only treatment for ground-nesting birds. Although spring burning may affect birds during their nesting period (Greenberg et al. 2007), this result in our study is somewhat confusing. First, the burn-only treatments were very light and not very effective in reducing fuels (Agee and Lolley 2006), tree density, or basal area (Harrod et al. 2007) or affecting the understory plant cover or species richness (Dodson et al. 2008). The burns occurred on April 25–27, before onset of nesting (usually in mid-May). In addition, the thin-burn treatment was much more intense, yet survival was high compared with survival rates for ground-nesters in other treatments. This finding leads one to wonder whether perhaps there was a predisposition for burn-only stands to have lower survival rates, yet pretreatment results show similar survival rates for ground-nesters across all stands (Gaines et al. 2009). Clearly, additional research is needed to address the impacts of spring burning on ground-nesting birds.

Large trees served important functions as both nesting and foraging habitats (Lyons et al. 2008) for cavity-nesters and as nesting habitats for foliage-nesters (we did not study foraging habitat by foliage-nesters). The importance of tree diameter in habitat selection by cavity-nesting birds is well documented (Lundquist 1988, Lundquist and Manuwal 1990, Adams and Morrison 1993, Bull et al. 1997, Weikel and Hayes 1999, Bevis and Martin 2002, Bunnell et al. 2002, Dickson et al. 2004). The retention of large tree habitat within our treatments provided important functions that reduced the negative effects of tree removal on many focal bird species. Restoration of large tree structures across dry forest landscapes will be important for reducing the decline of focal avian species such as the white-headed woodpecker (Altman 2000, Wisdom et al. 2000), western bluebird (Wisdom et al. 2000, Germaine and Germaine 2002), and possibly others.

Study Limitations

Our study area included a variety of plant associations within the broader category of dry forests. Although these forests have similar disturbance regimes, the variability among sample units could make treatment effects more difficult to detect. However, given the emphasis on dry forests for restoration and its relevancy to managers, our study provides a realistic measure of the diversity inherent in these forest types.

Our ability to estimate species-specific detection probabilities was limited by sample sizes for many species. However, we addressed this issue in three ways: we adjusted our density estimates to account for detection probabilities for those species with adequate samples sizes and where our analysis showed that an adjustment was needed; we evaluated detection probabilities for more common species at various radii to assess how much it varied among treatments; and we used the information on detection probabilities for more common species to identify a conservative radii (50 m) to truncate our data and limit any bias unequal detection might have caused for less common species.

Nur et al. (1999) recommended against pooling survival data across species because of potential differences in the length of nest period or different probabilities of daily survival. The FFS study nest monitoring protocol sampled only two of three replicates of each treatment and sample units (stands) were relatively small, both of which resulted in small sample sizes. In fact, sample sizes were too small (Hensler and Nichols 1981, Nur et al. 1999) to assess treatment effects on daily survival for individual species, and the small sample sizes posed limitations on the kind of analysis methods we could use. For long-term monitoring of the FFS sites we recommend that nest monitoring be performed on all of the 12 study stands.

The small sample sizes of nests limited our analysis of nesting substrate. However, we believe this limitation posed less of a problem than it did for survival as our results from the nesting habitat analysis are consistent with those of other studies and are biologically meaningful.

Management Implications

Much has been written about the need for active management of dry forests in the interior West to restore forest structure and composition and to reduce fuel loads. Studies, such as the FFS study, provide important information for managers to design effective treatments and better understand the effects of treatments on a variety of resources (Lehmkuhl et al. 2007b). Interdisciplinary results from our study area suggest that properly designed restoration treatments can be used to successfully reduce fuel loads (Agee and Lolley 2007), restore forest structure and composition (Harrod et al. 2007), and enhance habitat conditions for several focal avian species (Gaines et al. 2007).

Gaines et al. (2007) provided evidence that several focal avian species showed positive numerical responses to restoration treatments. Lyons et al. (2008) showed that restoration treatments enhanced foraging habitat conditions for bark-gleaning species. In this study we report generally neutral to positive numerical responses for several focal avian species and generally positive responses on the daily survival rates of most nesting guilds to restoration treatments (with the possible exception of spring burning on ground-nesting species). Collectively these studies provide information on numerical response, functional response, and survival response of avian species to restoration treatments. Based on this body of research we offer the following implications for managers and researchers to consider: (1) thinning from below followed by prescribed burning can be used as an effective tool to restore habitat for many avian focal species, including neotropical and migratory species; (2) spring burning may not have the desirable effect on restoration of habitat structure for focal avian species if conducted when conditions are too cool and moist; (3) large trees (and snags) in dry forests provide important habitat for foraging and nesting and are a key component in maintaining or restoring the viability of focal avian species; (4) the effects of spring burning on ground-nesting species needs more focused research with greater sample sizes to better understand the relationship between the timing and intensity of spring prescribed burns and effects on avian nesting and survival, and (5) although we have gained understanding of the short-term effects of restoration treatments on a variety of aspects of avian ecology, long-term research and monitoring are still needed to understand land-bird responses to these treatments.

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