EFFECTS OF RECENT BURNING ON BREEDING BIRD COMMUNITY STRUCTURE IN ASPEN FORESTS

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Abstract.—Fire is an important disturbance that stimulates aspen regeneration and expansion in the western United States. We investigated the impacts of burning aspen-dominated habitats on landbird community structure in the western foothills of the Wind River Range, Wyoming. Habitat structure, breeding bird species composition, richness, and relative abundance were compared in 1994 and 1995 between six independent pairs of sites, each pair consisting of a recently burned aspen site and a control site. Burned sites typically had lower canopy cover and depth, greater snag densities and aspen regeneration, and less shrub growth. We observed 45 diurnal woodland bird species overall. Western Wood-Pewees (*Contopus sordidulus*) and Warbling Vireos (*Vireo gilvus*) were detected significantly less often in the burned sites, while Mountain Bluebirds (*Sialia currucoides*) and Pine Siskins (*Carduelis pinus*) were detected significantly more often in the burned sites. Orange-crowned Warblers (*Vermivora celata*), Red-naped Sapsuckers (*Sphyrapicus nuchalis*), and chickadees (*Poecile* spp.) were significantly less abundant in the burned aspen when burn size or year was factored into the analysis. For many other commonly detected species, we had relatively low power to detect meaningful differences in abundance between recently burned and unburned aspen stands. Bird species richness was similar between treatments. Bird species composition differed consistently among burned-control pairs, but our data suggested that intersite variability in species composition in aspen is the norm in the absence of burning.

EFECTOS DE QUEMA RECIENTE EN LA ESTRUCTURA REPRODUCTIVA DE LA COMUNIDAD DE AVES EN BOSQUES DE *POPULUS*

Sinopsis.—El fuego es un importante factor de disturbios que estimula la regeneración y expansión de árboles de *Populus* en los Estados Unidos occidentales. Investigamos el impacto de quemar habitats dominados por los *Populus* en la estructura comunal de aves en las laderas occidentales de la Cordillera de Wind River en Wyoming. La estructura del habitat, la composición de especies en reproducción, la riqueza y la abundancia relativa de especies de aves se compararon en 1994 y 1995 entre seis pares de lugares independientes, cada par consistente de una localidad de *Populus* recién quemada y un área control. Las localidades quemadas tenían en general una cubierta y profundidad de dosel menor, mayor densidad de tocones y regeneración de árboles de *Populus*, y menor crecimiento de arbustos. Observamos un total de 45 especies de aves de bosque. Se detectaron significativamente menos *Contopus surdidulus* y *Vireo gilvus* en las áreas quemadas, mientras que *Sialia currucoides* y *Carduelis pinus* se detectaron signitivamente más a menudo en estas áreas. Aves de *Vermivora celata*, *Sphyrapicus nuchalis* y del género *Poecile* fueron significativamente menos abundantes en las áreas quemadas cuando el tamaño de la quema o el año fué un factor analizado. Tuvimos una potencia para detectar diferencias en abundancia de aves entre setos quemados y control relativamente pobre para muchas otras especies comunmente detectadas. La riqueza de especies fué similar entre tratamientos. La composición de especies difirieron considerablemente entre pares de áreas quemadas-control, pero nuestros datos sugieren que; a variabilidad en composición de especies entre lugares en *Populus* es la norma en la ausencia de la quema.

Forests dominated by trembling aspen (*Populus tremuloides*) provide

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habitat for a variety of bird species in the Rocky Mountain region (Salt 1957, Flack 1976, Winternitz 1976, DeByle 1981, Finch and Reynolds 1987, Turchi et al. 1995). Aspen is especially important because it is the only upland deciduous tree species that grows extensively in the central Rocky Mountains, and has been identified as a key habitat for wildlife in this region and elsewhere (Finch and Ruggiero 1993). Fire suppression efforts this century, in conjunction with heavy ungulate browsing, are thought to have allowed many western aspen forests to decline (Krebill 1972, Schier 1975, Gruell 1980, Mueggler 1989, Romme et al. 1995). Although western aspens form persistent communities (Reed 1971, Crawford et al. 1998), many are seral to coniferous and rangeland communities, with successional replacement occurring in as little as one aspen generation (Mueggler 1985). Almost all reproduction in aspen is vegetative, where adventitious shoots grow from lateral root systems in response to the death of the parent shoots (Barnes 1966). Natural disturbances such as disease, insects, and especially fire, stimulate aspen regeneration and expansion (Schier 1975, Bartos 1991).

Prescription burning is a method that is commonly used to stimulate aspen regeneration, because large areas can be effectively and inexpensively treated (Schier et al. 1985). To date, little is known about landbird community responses to burning aspen, despite the importance of fire to this forest type. The goal of our study was to examine the long-term impacts of burning in aspen forests on breeding bird community and habitat structure. Our objectives were to: (1) compare breeding bird community structure with respect to species composition, diversity, and relative abundance between burned and unburned aspen forests, and (2) simultaneously compare habitat conditions for birds in burned versus unburned aspen forests.

STUDY AREA AND METHODS

The study area was located in the western foothills of the Wind River Range on the Bridger-Teton National Forest, Sublette County, Wyoming (approx. 43°N, 110°W, elev. 2320–2750 m. Mature, even-aged aspen stands occupied a narrow band in the foothill physiographic region, usually in association with sagebrush (*Artemisia tridentata*) communities on south-facing slopes. Annual precipitation averaged 35 cm, and annual air temperature averaged 0 C (Martner 1986). The study area has been grazed since the 1850s, and is presently grazed from approximately June–September each year (range = 0.07–0.32 cow-calf pairs/ha) (B. Franklin, U.S. Forest Service, pers. comm.). There was no visual evidence of recent logging within the stands that we sampled.

Five recently burned aspen sites were located in 1994, and an additional burned site was added in 1995. These were the only available aspen-dominated areas that had been recently burned within the study area. Burned sites were variable in terms of year burned (1988–1993), aspen forest cover and patchiness, wetland abundance, topography, stand age, and fire coverage and intensity. Two sites were wildfire burns (1988), while the

remaining four sites were prescription burns. About 61% (range = 27–91%) of each burned site was moderately to severely burned, while the remaining pockets were either not exposed to fire or showed only minor evidence of burning, usually in the form of charred woody debris. Burned sites ranged from 17–166 ha in aspen forest cover (38–407 ha in total area). Each burned site was matched with a nearby control site of comparable area; selection was based on similarities in aspen forest cover, topography, and wetland abundance. The burned-control site pair served as the experimental unit in our analyses.

We measured avian and habitat characteristics from stations randomly selected on each site. All stations were in the aspen forest type, and were separated from each other by 150–500 m to ensure that most observations during bird surveys were independent (Manuwal and Carey 1991). Six to 20 stations were selected according to site area to characterize each site. Each member of a burned-control site pair had an equal number of stations.

Bird surveys.—Bird populations were sampled using fixed-radius point counts (Hutto et al. 1986). Surveys were conducted from 28 May–1 Jul. 1994 and from 30 May–9 Jul. 1995. Each survey station was visited three times each year, with each visit separated by at least one week. The order in which stations were surveyed was reversed on subsequent visits.

Surveys were conducted for an 8-min period, beginning immediately upon arrival at each station. Bird surveys began 15–30 min following local sunrise and typically lasted until 0930–1000 h MDT. Surveys were not conducted during periods of inclement weather, including periods of rain or snow, or when winds exceeded 19 km/h (Robbins 1981). During each 8-min point count, the number of individuals detected for each species within a 50-m radius was recorded. Bird locations were mapped within 50 m during each point count survey (Ralph et al. 1993), which minimized the probability of counting an individual more than once. We restricted our observations to diurnal, woodland species, and birds that were detecting flying over or through a station were not counted. However, birds circling above the canopy were considered valid detections.

Habitat measurements.—Habitat characteristics were measured in 1994 and 1995 at each survey station following James and Shugart (1970), with modifications provided by Noon (1981). Topographic data such as slope and elevation were measured at the center of each station. Vegetative characteristics were measured within seven 0.04-ha plots, nested within 50 m of the station center. Diameter at breast height (dbh), tree height, and canopy depth were measured for sampled trees (live or dead) ≥3 cm dbh. We categorized each tree stem into five decay stages, according to Thomas et al. (1979). Percent canopy cover (overstory and subcanopy) and ground cover attributes were estimated using the point intercept method. The overstory represented the highest layer of aspen canopy development. Subcanopy cover was estimated where a lower canopy developed in uneven-aged stands. Ground intercept readings were identified, measured for height, and grouped into the following lifeform cate-

gories: shrubs, forbs, graminoids, litter (leafy material, twigs), bare
ground, rocks, and woody debris (logs and tree branches). Aspen sucker
density and height were estimated for stems measuring <3 cm dbh. We
used the point-quarter technique to index downed log abundance in each
quarter plot by recording the distance to the nearest downed log ≥8 cm
in maximum diameter within an 11.3-m radius. All live conifers ≥10 cm
dbh were counted and tallied by species within 50 m of the station center.

Distances to the nearest conifer-dominated forest and sage/grassland
habitat (>1 ha) were measured from the center of each station, either in
the field or using aerial photographs. Field sketches were used to estimate
percent cover of each plant community type (e.g., aspen, sagebrush, wil-
low [*Salix* spp.] riparian) within 50 m of each station center.

Data analysis.—We estimated relative densities for each bird species at
each site by summing the maximum number of detections of the three
visits recorded at each station by year, and dividing by the number of
stations in the site. Species richness was calculated for each site by sum-
ming the number of species detected within each site per year. Species
richness was also characterized on a per station basis, using the mean
number of species detected per station. Species composition was com-
pared for each pair of burned-control sites using the Sørensen coefficient
of community similarity (Brower and Zar 1984), which yields a percent
similarity in bird species composition among matched sites. We also com-
pared species composition among unrelated sites randomly selected with-
in treatments as a basis for comparison.

To test for the effects of burning, bird survey and habitat data collected
from stations were collapsed into a single measurement per site, using
either means for metric data or proportions for point-intercept data. Hab-
itat parameters were tested between treatments using the Wilcoxon
signed-ranks test (Zar 1984). We tested bird species abundance and rich-
ness data for treatment effects, year effects, and treatment-year interac-
tions using multivariate analysis of variance with a repeated measures de-
sign (Norusis 1990). Power analyses were conducted for the bird survey
data averaged across years as described in Green (1989).

Using the Spearman rank correlation coefficient, we correlated year of
burn, burn size, and burn severity with the difference among treatment
pairs for richness and abundance parameters previously estimated (burn$_i$
− control$_i$). Year from burn ranged from 2–7 yr between time of treat-
ment and year surveyed, and burn severity was characterized by estimating
the proportion of stations at each site that had been moderately to se-
verely burned. In recognition of our small sample size ($n = 5$ treatment-
control pairs over both years), this approach allowed us to incorporate
these variables into the analysis.

All statistical tests were conducted using SPSS/PC+ ver. 4.01 (SPSS Inc.
1991). All tests were considered significant at a P-value < 0.10. Using a
relatively large α-level increased the probability of a type I error, but de-
creased the probability of committing a type II error. We feel this is more
prudent when the assessing the effects of public land management prac-

tices on wildlife populations and their habitats (Askins et al. 1990, Thompson et al. 1992).

<center>RESULTS</center>

Forty-five diurnal woodland species were documented from all sites over both years; 38 were recorded from the burned and 34 from the control sites. The three most commonly detected species accounted for nearly 47% of all observations: 21% Dusky Flycatchers, 15% Warbling Vireos, and 11% House Wrens (scientific names are provided in Table 1). We detected fewer Western Wood-Pewees and Warbling Vireos in the burned sites, and more Mountain Bluebirds and Pine Siskins in the burned sites over both years (Table 1). No significant difference in mean abundance was detected for any species between years. We observed significant interactions between burning and year for Northern Flicker, Red-naped Sapsucker, and Brown-headed Cowbird relative abundance. Northern Flickers had significantly more observations in the control sites in 1995 ($SE_d = 0.22$, df = 5, $P = 0.08$), but not in 1994 ($SE_d = 0.03$, df = 4, $P = 0.69$). Red-naped Sapsuckers were detected twice as often in the burned sites in 1995 ($SE_d = 0.16$, df = 5, $P = 0.06$), but detected evenly among treatments in 1994 ($SE_d = 0.0$, df = 4, $P = 0.98$). We did not find any significant differences between treatments for Brown-headed Cowbird abundance for either year separately, although we observed more in the burned sites than in the controls in 1994, while the opposite was true in 1995 ($F_{int} = 7.33$, df = 1, $P = 0.05$).

Bird species richness averaged 15.8 in 1994 and 18.0 in 1995 per burned site and 16.8 species in 1994 and 18.7 species in 1995 per control site. Over both years combined the control sites averaged 1.7 ± 1.4 [SE_d] more bird species, although the difference was not significant ($F = 1.49$, $P = 0.289$). We also obtained similar richness values between treatments when estimated per survey station ($SE_d = -0.33$, $SE_d = 0.90$, $P = 0.345$). Community similarity in species composition (Sørensen coefficient) between burned-control paired sites averaged $72 \pm 4\%$ [SE] in 1994 ($n = 5$) and $67 \pm 4\%$ in 1995 ($n = 6$). Among randomly paired sites within treatment type, we obtained community similarity values of $70 \pm 5\%$ in 1994 and $59 \pm 4\%$ in 1995.

We detected a strong, negative relationship between the number of years since burning and the paired difference between burned and control site abundance for Mountain and Black-capped Chickadees combined ($r_s = -0.975$, $n = 5$, $P = 0.005$)(Fig. 1). We detected significant relationships between size of burn and the differences between burned and control site abundance for Orange-crowned Warblers and Red-naped Sapsuckers. As site size increased, the average number of Orange-crowned Warbler detections decreased in the burned sites ($r_s = -1.000$, $P < 0.001$), while the number of Red-naped Sapsucker detections increased in the burned sites ($r_s = 0.975$, $P = 0.005$)(Fig. 1).

There were marked differences in several habitat parameters between burned and unburned aspen sites, particularly those related to aspen fo-

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TABLE 1. Comparison of bird counts (number per 50-m radius point count) between burned and unburned sites, for 1994 and 1995 survey data combined. Power $(1 - \beta)$ was estimated for a 25% difference from the unburned estimate at $\alpha = 0.10$.

Species	Mean			F	P	Power (%)
	Burned	Control	Differ. (SE_d)			
Red-naped Sapsucker (Sphyrapicus nuchalis)	0.13	0.22	-0.09 (0.05)	3.66	0.128[a]	21.7
Hairy Woodpecker (Picoides villosus)	0.04	0.06	-0.02 (0.04)	0.40	0.564	9.3
Northern Flicker (Colaptes auratus)	0.39	0.28	0.11 (0.08)	2.35	0.200[a]	16.3
Western Wood-Pewee (Contopus sordidulus)	0.16	0.56	-0.40 (0.15)	7.18	0.055	17.4
Dusky Flycatcher (Empidonax oberholseri)	1.29	1.51	-0.22 (0.12)	3.43	0.138	80.1
Warbling Vireo (Vireo gilvus)	0.88	1.33	-0.45 (0.18)	6.55	0.063[a]	36.2
Tree Swallow (Tachycineta bicolor)	0.53	0.41	0.12 (0.14)	0.83	0.413	14.2
Poecile spp.[b]	0.07	0.20	-0.13 (0.10)	1.68	0.264	10.5
House Wren (Troglodytes aedon)	0.78	0.83	-0.05 (0.11)	0.28	0.627	37.1
Mountain Bluebird (Currucoides mexicanus)	0.50	0.27	0.23 (0.08)	8.77	0.042	16.9
American Robin (Turdus Migratorius)	0.31	0.38	-0.07 (0.09)	0.68	0.456	19.4
Orange-crowned Warbler (Vermivora celata)	0.13	0.24	-0.11 (0.08)	2.08	0.223	14.4
Yellow Warbler (Dendroica petechia)	0.22	0.13	0.09 (0.07)	1.73	0.259	10.7

TABLE 1.   Continued.

Species	Mean		Differ. (SE_d)	F	P	Power (%)
	Burned	Control				
Yellow-rumped Warbler (*Dendroica coronata*)	0.10	0.14	−0.04 (0.07)	0.27	0.628	10.3
Green-tailed Towhee (*Pipilo chlorurus*)	0.30	0.25	0.05 (0.14)	0.12	0.748	10.0
Chipping Sparrow (*Spizella passerina*)	0.11	0.13	−0.03 (0.03)	0.78	0.427	21.7
White-crowned Sparrow (*Zonotrichia leucophrys*)	0.18	0.19	−0.01 (0.10)	0.02	0.900	10.7
Dark-eyed Junco (*Junco hyemalis*)	0.64	0.84	−0.20 (0.14)	2.30	0.204	30.3
Lazuli Bunting (*Passerina amoena*)	0.08	0.06	0.02 (0.09)	0.07	0.807	6.9
Brown-headed Cowbird (*Molothrus ater*)	0.37	0.27	0.10 (0.10)	1.05	0.364[a]	13.4
Cassin's Finch (*Carpodacus cassinii*)	0.10	0.03	0.07 (0.04)	2.06	0.225	7.2
Pine Siskin (*Carduelis pinus*)	0.40	0.07	0.33 (0.11)	8.84	0.041	6.8
Species Richness/site	16.5	18.2	−1.7 (1.4)	1.49	0.289	74.8
Species richness/station	5.9	6.2	−0.3 (0.9)	0.66	0.463	89.1

[a] Indicates a significant ($P < 0.10$) interaction between year and treatment. There were no significant year effects.
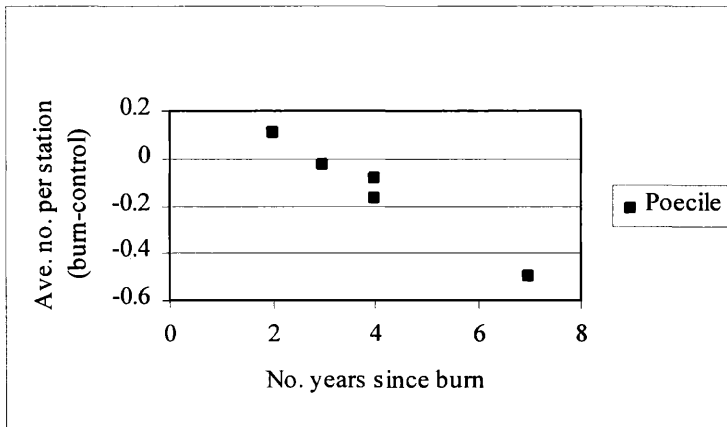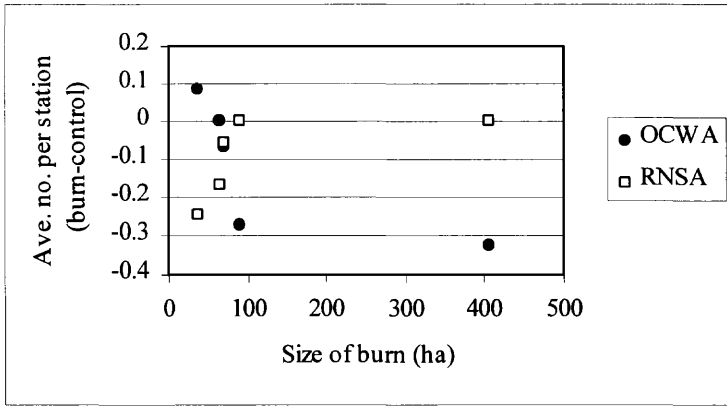[b] Black-capped (*P. atricapillus*) and Mountain (*P. gambeli*) Chickadee detections combined.

FIGURE 1. Relationship between burn size, burn year, and the paired difference (burn$_i$ − control$_i$) in relative density for *Poecile* spp., Orange-crowned Warblers (OCWA), and Red-naped Sapsuckers (RNSA). Number of years since burned represents the difference between year burned and last year surveyed.

liage and regeneration, snag density, and shrub development (Table 2). We checked for potential biases in control-site selection by comparing several physiographic and landscape factors that would be largely unaffected by recent burning. We found no substantial mean differences between burned-control pairs in aspen forest cover (3.3 ± 3.7 ha [SE], slope (2.5 ± 1.8°), or elevation (3 ± 39 m). Modest differences in average distance to coniferous forest (68 ± 88 m), and sage/grassland communities (30 ± 14 m) were estimated. On average, 50% of the survey stations in the burned sites had standing water within 100 m of each survey station's center, as compared with 52% (SE_d = 8%) in the controls. Canopy structure

TABLE 2. Comparison of habitat characteristics between burned and unburned aspen sites ($n = 6$).

Habitat variable	Mean			
	Burned	Unburned	Differ. (SE)	P
Aspen Canopy				
Canopy height (m)	9.0	9.4	−0.4 (0.9)	0.753
Canopy cover (%)	18.3	46.9	−28.6 (5.8)	0.027
Canopy depth (m)	4.1	4.6	−0.5 (0.3)	0.074
Subcanopy (%)	0.6	1.2	−0.6 (0.4)	0.345
Aspen Stem Density (ha)				
Live (≥3 cm dhb)	651	1082	−431 (285)	0.284
Snag (≥3 cm dbh)	616	230	386 (166)	0.075
Live (≥10 cm dbh)	250	500	−250 (108)	0.046
Snag (≥10 cm dbh)	191	99	92 (42)	0.075
Downed Aspen Logs				
Maximum diameter (cm)	12.9	13.5	−0.6 (0.4)	0.463
Length (m)	6.6	7.2	−0.6 (0.8)	0.250
Relative frequency[a]	84.4	92.8	−8.4 (4.0)	0.043
Aspen Sucker Density (ha)				
<1 cm dbh	18,288	2495	15,793 (5487)	0.046
1–2 cm dbh	575	195	380 (355)	0.225
2–3 cm dbh	180	149	31 (149)	0.893
Conifers (≥10 cm dbh)				
Density (ha)	0.9	1.3	−0.4 (0.7)	0.500
Species richness	1.8	2.0	−0.2 (0.3)	0.593
Understory Cover (%)				
Bare ground	4.3	1.5	2.8 (1.8)	0.249
Shrub cover	8.6	19.6	−11.0 (2.2)	0.028
Forb	27.6	26.6	1.0 (4.4)	0.753
Graminoid	34.2	27.5	6.7 (3.2)	0.075
Berry producers	8.8	17.6	−8.8 (2.2)	0.028
Litter	7.8	10.6	−2.8 (2.7)	0.345
Woody debris	5.2	5.8	−0.6 (0.4)	0.345
Understory Height (cm)				
Shrub	33.5	44.0	−10.5 (1.8)	0.028
Forb	31.9	20.1	1.8 (1.7)	0.463
Graminoid	26.3	24.6	1.7 (2.5)	0.600

[a] Percent occurrence among sample plots per site.

in either treatment was generally single layered, indicative of even-aged aspen growth throughout our sites. Standing aspen density (live or dead) ≥3 cm dbh was also similar between treatments, averaging 1268 stems/ha in the burned sites, and 1312 stems/ha in the control sites ($SE_d = 330$).

DISCUSSION

Burning aspen forests, whether seral or climax, essentially reverses succession and returns these communities to an earlier seral stage, usually

dominated by young aspen regeneration (Bradley et al. 1992). However, the final result is not consistent, since the outcome of a burning event is largely a function of the magnitude and distribution of aspen mortality, as well as the initial health of the clones. However, we can at least suggest some patterns that were evident from our data. Burning clearly affected several habitat components important for birds; especially those related to aspen canopy, reproduction, and downed log abundance, shrub development, and snag density. The effects were not universal and were variable in magnitude among our sites, which can be attributed in part to how burns vary in size, intensity, season, and coverage (Rotenberry et al. 1995).

We observed differences in relative abundance for several species. We feel that we can logically attribute the effects of burning on bird abundance in part to habitat alteration, and not to the actual fire events themselves (Bendell 1974). Two of the salient habitat alterations found in this study were related to the canopy structure of the burned sites. Both canopy closure and depth were diminished in patches and over large areas, depending on burn severity and coverage. Reductions in canopy foliage probably explain why we recorded fewer Warbling Vireos and Western Wood-Pewees in the burned sites, where the nesting and foraging requirements for these species are principally met in the canopy (James 1971, Verbeek 1975, Whitmore 1975, James 1976). Similar patterns were observed for Warbling Vireos on an aspen clearcut in Utah (DeByle 1981). Raphael et al. (1987) found that canopy nesters were collectively half as abundant on a burned, forested plot in the Sierra Nevada. We failed to find differences in abundance for other canopy nesting species observed in this study, with the possible exception of Pine Siskins. Although not commonly detected, Pine Siskins were considerably more abundant in the burned sites, and only rarely detected in the controls. We observed little territorial activity and suspect that they used these sites only for foraging.

Cavity-nesting species composed a significant portion of the bird community in this study (27%). Although the burn sites had more snags as expected, it is not clear that the cavity nesters as a group benefited, suggesting that the availability of suitable nest sites are not limiting in unburned, mature aspen forests. Red-naped sapsuckers were the only primary cavity nester that showed any differences in abundance between treatments, although the effect was confounded by burn size. As the size of the burn increased, the relative abundance of Red-naped Sapsuckers approached that of the controls, suggesting that their populations were negatively impacted in the smaller burns. We also found no effect in abundance for secondary cavity nesters as a group. *Poecile* species were collectively more prevalent in the control sites, especially as the interval between year burned and year surveyed increased. Reasons for this are unclear, but chickadees may be sensitive to the steady loss of canopy foliage as trees die following burning.

Mountain Bluebirds were the only cavity-nesting species that was more abundant in the burned sites, perhaps benefiting from greater snag avail-

ability. Mountain Bluebirds inhabit a variety of open and ecotonal wood-
lands, and tend to benefit from management activities that create forest
edge, and control shrub development in the understory (Power and Lom-
bardo 1996). However, most of our stands were naturally fragmented, and
the burns did not create forest edge, although shrub cover and height
were significantly reduced in the burned sites. Shrub-dependent species,
particularly Orange-crowned Warblers and Dusky Flycatchers, were ob-
served more often in the unburned aspen, although the effect was pos-
sibly mitigated somewhat by the large production of root suckers observed
in the burns, which may have served as surrogate shrubs, under dense
growing conditions.

Despite the apparent effects of burning on key aspects of the habitat
structure, we observed little overall difference in bird species richness
between burned and unburned aspen sites, although we had adequate
power to do so (Table 1). We can at least conclude that there was no
large-scale reduction in avian diversity in response to burning, as indexed
by species richness. We also found little evidence that species composition
was affected by these burning events. Although average species compo-
sition consistently differed among matched treatment-control pairs, ran-
domly matched sites within treatment showed similar differences. If there
had been a relationship or common factor(s) that distinguished burned
aspen forests in avian community composition, the percentage of species
shared among like-treatment pairs should have increased on average. Our
data instead suggested that inter-site variability in bird species composi-
tion is the norm in the absence of burning.

Our data showed that many aspects of landbird community structure
were not substantially affected by recent burning in aspen forests, which
has been a noted pattern in other forested habitats subjected to fire (Bock
and Lynch 1970, Emlen 1970, Bendell 1974, Blake 1982, Rotenberry et
al. 1995). Although we were able to detect differences in abundance for
several species, there was little evidence of large-scale changes in distri-
bution or abundance. None of the commonly detected species were re-
stricted to only burned or unburned aspen stands. Our data suggest that
the loss of aspen stands through succession is probably more detrimental
to birds associated with aspen communities than by treating such stands
using prescription burns. However, we feel that the topic of burning in
aspen forests and its implications for landbirds warrants continued explo-
ration. We demonstrated that the population size of some species were
affected, but we had relatively low power to explore the consequences of
burning on other species that are commonly associated with Rocky Moun-
tain aspen. Prescription burning is an important management tool used
to regenerate aspen in western forests. There are still unanswered ques-
tions regarding the relative effects of burn size, intensity, frequency and
season on aspen habitat and bird community structure.

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