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VARIATION IN FIRE REGIMES OF THE ROCKY MOUNTAINS: IMPLICATIONS FOR AVIAN COMMUNITIES AND FIRE MANAGEMENT

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Abstract. Information about avian responses to fire in the U.S. Rocky Mountains is based solely on studies of crown fires. However, fire management in this region is based primarily on studies of low-elevation ponderosa pine (*Pinus ponderosa*) forests maintained largely by frequent understory fires. In contrast to both of these trends, most Rocky Mountain forests are subject to mixed severity fire regimes. As a result, our knowledge of bird responses to fire in the region is incomplete and skewed toward ponderosa pine forests. Research in recent large wildfires across the Rocky Mountains indicates that large burns support diverse avifauna. In the absence of controlled studies of bird responses to fire, we compared reproductive success for six cavity-nesting species using results from studies in burned and unburned habitats. Birds in ponderosa pine forests burned by stand-replacement fire tended to have higher nest success than individuals of the same species in unburned habitats, but unburned areas are needed to serve species dependent upon live woody vegetation, especially foliage gleaners. Over the last century, fire suppression, livestock grazing, and logging altered the structure and composition of many low-elevation forests, leading to larger and more severe burns. In higher elevation forests, changes have been less marked. Traditional low-severity prescribed fire is not likely to replicate historical conditions in these mixed or high-severity fire regimes, which include many mixed coniferous forests and all lodgepole pine (*Pinus contorta*) and spruce-fir (*Picea-Abies*) forests. We suggest four research priorities: (1) the effects of fire severity and patch size on species' responses to fire, (2) the possibility that postfire forests are ephemeral sources for some bird species, (3) the effect of salvage logging prescriptions on bird communities, and (4) experiments that illustrate bird responses to prescribed fire and other forest restoration methods. This research is urgent if we are to develop fire management strategies that reduce fire risk and maintain habitat for avifauna and other wildlife of the Rocky Mountains.

Key Words: coniferous forests, fire management, fire regimes, passerine birds, U.S. Rocky Mountains, woodpeckers.

VARIACIÓN EN REGÍMENES DEL FUEGO EN LAS ROCALLOSAS: IMPLICACIONES PARA COMUNIDADES DE AVES Y MANEJO DEL FUEGO

Resumen. La información respecto a las respuestas de las aves al fuego en las Rocallosas de los Estados Unidos, está basado únicamente en estudios de incendios de copa. Sin embargo, el manejo de incendios en esta región esta basada primordialmente en estudios de bosques de pino ponderosa (*Pinus ponderosa*) de baja elevación, los cuales se mantienen primordialmente con incendios en la primera capa vegetativa. En contraste a ambas tendencias, la mayoría de los bosques de las Rocallosas están sujetas a regimenes mixtos de severidad de incendios. Como resultado, nuestro conocimiento de las respuestas de las aves a los incendios en la región es incompleta y dirigida hacia los bosques de pino ponderosa. Recientes investigaciones de grandes incendios en las Rocallosas, indican que grandes incendios ayudan a la avifauna. En la ausencia de estudios controlados en las respuestas de las aves al fuego, utilizando resultados de estudios en habitats incendiados y sin incendiar, comparamos el éxito reproductivo de seis especies que anidan en cavidades. Aves en bosques de pino ponderosa quemado por incendios de reemplazo, tienden a obtener un mayor éxito de anidación que los individuos de la misma especie en habitats sin quemar, pero se necesitan áreas sin quemar, que sirvan a especies dependientes de vegetación forestal viva, especialmente de follaje espigado. Desde el último siglo, la supresión de incendios, el pastoreo y los aprovechamientos forestales han alterado la estructura y composición de varios bosques de baja elevación, llevándolos a incendios mayores y severos. En bosques con mayor elevación, los cambios han sido menos marcados. Es muy poco probable replicar condiciones históricas en estos regimenes mixtos y de alta severidad con quemas prescritas tradicionales de baja severidad , las cuales incluyen varios bosques de coníferas y todos los bosques de pino (*Pinus contorta*) y de abeto (*Picea-Abies*). Sugerimos cuatro prioridades de investigación: (1) efectos de la severidad del incendio y tamaño del parche, en las respuestas de la especie al fuego, (2) la posibilidad de que bosques después de un incendio sean fuentes efímeras para algunas especies de aves, (3) los efectos de incendios prescritos en aprovechamientos forestales de salvamento en comunidades de aves, y (4) experimentos que ilustren respuestas de aves a incendios preescritos y otros métodos de restauración forestal. Esta investigación es urgente si queremos desarrollar estrategias de manejo del fuego, las cuales reduzcan el riesgo de incendios y mantengan el habitat para la avifauna y otras especies silvestres de las Rocallosas.

Forest landscapes of the U.S. Rocky Mountains are structured by a complex interplay of climate, topography, soils, and disturbance (Peet 2000, Schoennagel et al. 2004). They are shifting mosaics whose vegetation reflects variation in disturbance frequency, severity, and time since disturbance, which ranges from years to centuries (Peet 2000). Many of these fire regimes have been altered since Euro-American settlement due to fire suppression, logging, livestock grazing, and, in some cases, climate change (Veblen 2000, Allen et al. 2002, Schoennagel et al. 2004). After decades of fire suppression, elevated fuel loads in many forests have increased the likelihood of unusually large and severe fires (Arno and Brown 1991, Covington and Moore 1994), and the yearly area burned has increased (Grissino-Mayer and Swetnam 2000, Keane et al. 2002).

Severe wildfire seasons in 2000 and 2002 (collectively, 6,800,000 ha burned) focused public attention on the risks posed by fuel accumulations (Graham et al. 2004), and served as an impetus for the National Fire Plan (USDA 2000) and the Healthy Forests Initiative (White House 2002). This initiative was passed into law as HR1904, the Healthy Forests Restoration Act of 2003. A primary goal of these federal programs is to diminish the risk of severe wildland fire by reducing fuel loads and restoring historical forest structure and fire regimes. Prescribed fire and mechanical treatments are increasingly being used to meet this goal.

An assumption driving the recent fire management initiatives is that by reproducing the range of forest conditions and fire regimes that characterized a specific location and time period, we will provide the myriad ecological conditions that a diverse array of species require (e.g., Covington et al. 1997, Keane et al. 2002, Graham et al. 2004). However, the ecological paradigm underlying recent fire management policies in many Rocky Mountain forests, namely frequent understory fires and open forest structures (Covington and Moore 1994, Swetnam et al. 1999, Allen et al. 2002), was developed primarily from experience in ponderosa pine forests of the American Southwest (see Ehle and Baker 2003, Schoennagel et al. 2004). Recent evidence, however, suggests that historical fire regimes and forest structures of ponderosa pine forests were considerably more variable than suggested by the southwest paradigm (Brown and Sieg 1996, Shinneman and Baker 1997, Brown et al. 1999, Veblen et al. 2000). Thus, Rocky Mountain species associated with crown-burned forests, such as Lewis's Woodpeckers (*Melanerpes lewis*) and Black-backed Woodpeckers (*Picoides arcticus*), may be negatively affected by the southwest paradigm's emphasis on understory fire (Dixon and Saab 2000, Saab and Vierling 2001).

Managers who oversee Rocky Mountain forests require a fuller understanding of the variability inherent in the region's fire regimes, as well as the responses of its avifauna along such range of variation. In this paper, we summarize these topics. First, we review current knowledge about historical fire regimes for five dominant forest types. We discuss the degree to which fire regimes have been altered since Euro-American settlement. For each forest type, we summarize studies that have investigated the response of birds to wildfires and fire exclusion. Finally, we discuss the implications of forest restoration and fire management programs for avian communities of the Rocky Mountains.

ROCKY MOUNTAIN FORESTS

For purposes of this review, we define the U.S. Rocky Mountain region as the area from northern Montana and Idaho southward across the interior West, through Wyoming and Colorado to northern New Mexico (Fig. 1). Our definitions and descriptions of major vegetation types of the Rocky Mountains are taken largely from Peet (2000) and Arno (2000).

We describe five major vegetation types in this review: (1) pinyon-juniper (*Pinus-Juniperus*) woodland, (2) ponderosa pine forest, (3) mixed-coniferous forest, (4) lodgepole pine forest, and (5) spruce-fir forest. These vegetation classifications are derived from gradients in elevation, moisture, substrate, and disturbance regime (Peet 2000).

For each vegetation type we describe the distribution, elevation, dominant plant species, and characteristic birds, including those identified as priority species by Partners in Flight (2004). We also describe fire regimes for each vegetation type prior to and after European settlement, alterations to fire regimes, and probable effects on birds.

Floristically, the Rocky Mountains can be divided into several regions, two within our area of interest: the southern Rocky Mountains, from southern Colorado to central Wyoming, and the central Rocky Mountains of central Wyoming to Jasper National Park, Canada (Peet 2000). Across these regions, forest vegetation ranges from low elevation, dry forests to high elevation, mesic forests with various fire regimes (Fig. 1, 2; Peet 2000, Schmidt et al. 2002). Forest cover types occur from 1,100–3,500 m (limits vary geographically), and annual precipitation ranges from 12–245 cm. We used current cover types

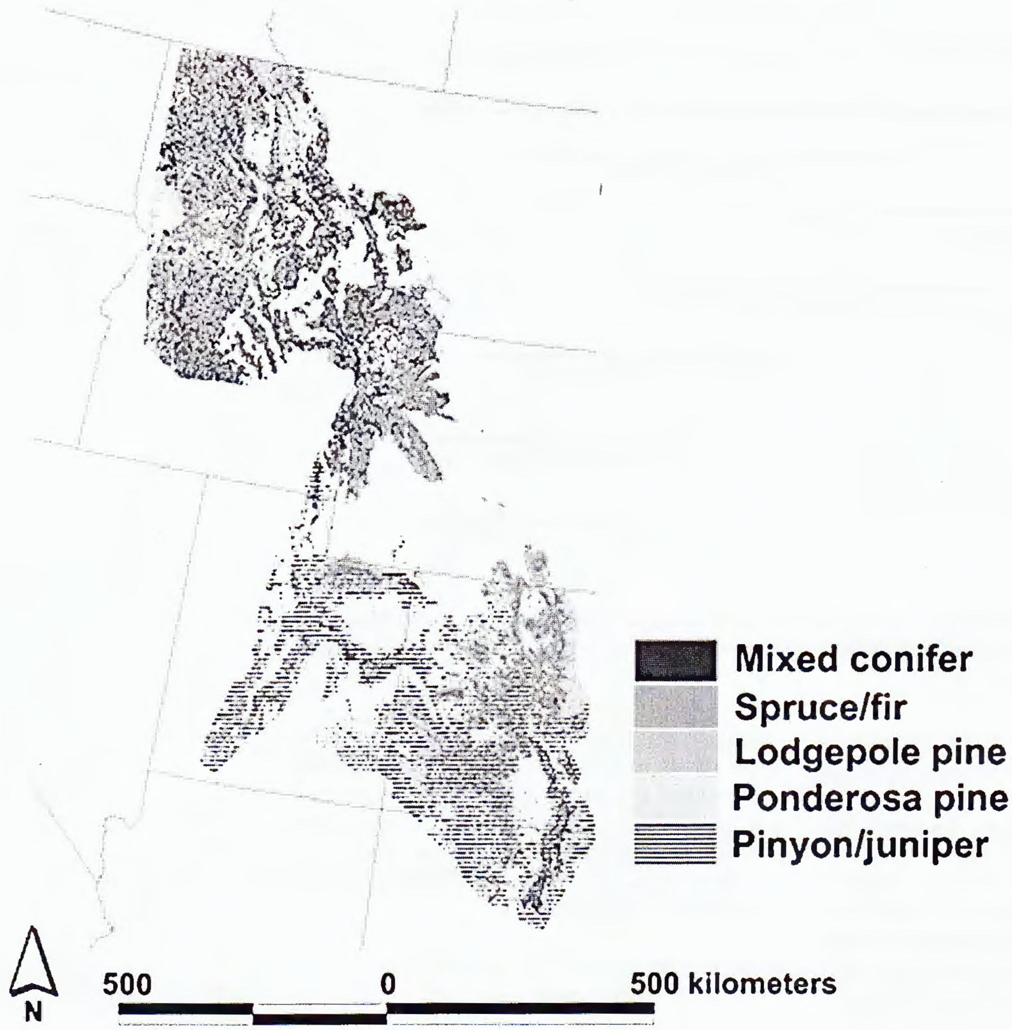

FIGURE 1. Map of current forest cover types in the U.S. Rocky Mountains (taken from Schmidt et al. 2002).

mapped by Schmidt et al. (2002) to estimate the area (in million ha) occupied by each of five major vegetation types within the U.S. Rocky Mountains: (1) pinyon-juniper woodland, 5.0; (2) ponderosa pine forest, 5.6; (3) mixed-coniferous forest, 8.7; (4) lodgepole pine forest, 9.7; and (5) spruce-fir forest, 5.0.

PINYON-JUNIPER WOODLANDS

Pinyon-juniper (pygmy) woodlands are most prevalent in the Madrean and southern Rocky Mountains (Peet 2000). West of the continental divide, pinyon-juniper woodlands extend northward into Idaho (Daubenmire 1943). Pinyon pine (*Pinus edulis*) occurs throughout the range; one-seed juniper (*Juniperus monosperma*) occurs on the eastern slope, whereas singleleaf pinyon (*Pinus monophylla*) and Utah juniper (*Juniperus. osteosperma*) share dominance with pinyon pine on the western slope (Daubenmire 1943). Rocky Mountain juniper (*Juniperus scopulorum*) is co-dominant with Utah juniper over much of the southern Rocky Mountains, and is frequent in the pinyon zone and adjacent lower reaches of ponderosa pine woodlands (Peet 2000). Stand densities tend to increase with moisture and elevation (Paysen et al. 2000).

The role of fire in these habitats remains poorly understood (Baker and Shinneman 2004). Frequent

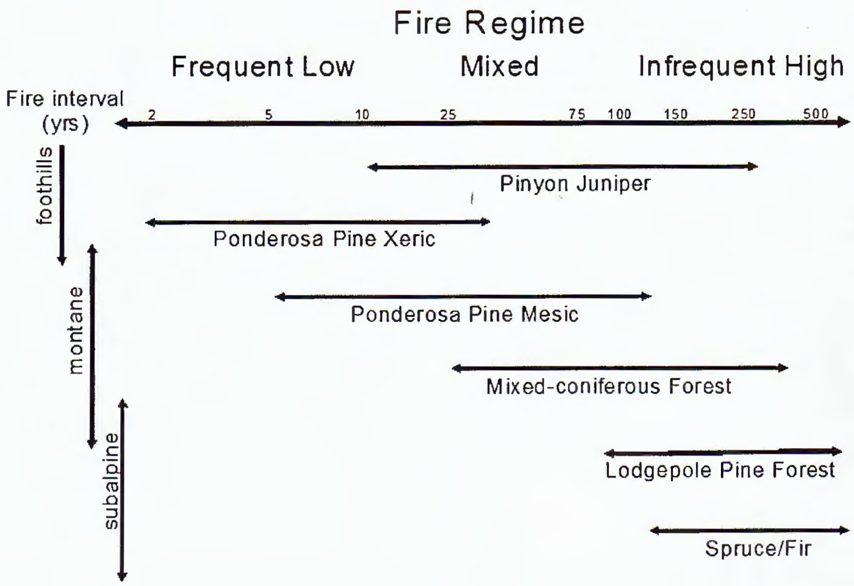

FIGURE 2. Range of variation in historical fire regimes for dominant forest types of the U.S. Rocky Mountains. Information for this graph was based largely on Arno (2000) and Schoennagel et al. (2004), and other sources referenced in the text by dominant forest type.

surface fires at intervals from 10 to <35 yr were considered prevalent in pinyon-juniper woodlands of the Rocky Mountains (e.g., Paysen et al. 2000). Recent evidence, however, suggests that natural fires in dense stands were infrequent and severe, occurring at intervals of 200–300 yr or longer (Floyd et al. 2000, Romme et al. 2003, Baker and Shinneman 2004). Frequent, low-severity fires were probably more common in the upper ecotone than in the closed woodland zone of pinyon-juniper forests (Baker and Shinneman 2004). A clear understanding of historical fire regimes at both local and landscape scales is sorely needed.

BIRDS OF PINYON-JUNIPER WOODLANDS

Characteristic birds of pinyon-juniper woodlands include Ferruginous Hawk (*Buteo regalis*), Gray Flycatcher (*Empidonax wrightii*), Ash-throated Flycatcher (*Myiarchus cinerascens*), Gray Vireo (*Vireo vicinior*), Western Scrub-Jay (*Aphelocoma californica*), Pinyon Jay (*Gymnorhinus cyanocephalus*), Juniper Titmouse (*Baeolophus ridgwayi*), Bushtit (*Psaltriparus minimus*), Blue-gray Gnatcatcher (*Polioptila caerulea*), Black-throated Gray Warbler (*Dendroica nigrescens*), and Virginia's Warbler (*Vermivora virginiae*) (Balda and Masters 1980). Partners in Flight (2004) priority bird species for this habitat include Gray Flycatcher, Gray Vireo, Pinyon Jay, and Juniper Titmouse. Many of these species are pinyon-juniper obligates (e.g., Juniper Titmouse), and all of these species rely on pinyon-juniper as their primary breeding habitat.

BIRD RESPONSE TO FIRE IN PINYON-JUNIPER WOODLANDS

To our knowledge, no detailed information is available on avian response to fire in pinyon-juniper woodlands in the Rocky Mountains. Response of vegetation and birds to fire will likely depend upon prefire plant composition and successional stage (Miller and Tausch 2001). Depending on fire severity, the loss of cover for shrub and tree-nesting species such as Bushtit, Gray Flycatcher, and Black-throated Gray Warbler may initially result in a negative response by these species. Residual snags would likely provide nest sites for cavity-nesting species such as Western (*Sialia mexicana*) and Mountain Bluebirds (*Sialia currucoides*). Site-specific studies are needed to evaluate these possibilities given the range of variability in fire regimes that likely exists in this habitat.

National assessments suggest that many pinyon-juniper woodlands have missed one or more low-severity surface fires since Euro-American settlement (Baker and Shinneman 2004). For these reasons, low-severity, prescribed fire has been the focus of fire management in pinyon-juniper woodlands. This management emphasis may not be appropriate throughout these woodlands, and many of the pinyon-juniper forests were likely maintained by infrequent, high-severity fire (Baker and Shinneman 2004).

Disproportionate attention on low-severity surface fire, or treatments that create like conditions, could adversely affect avian species associated with mature pinyon-juniper woodlands (cf. Horton 1987, Sedgwick 1987). Nesting numbers of Virginia's Warblers declined after applications of prescribed fire in ponderosa pine woodlands, possibly due to removal of nesting sites in low shrubs and understory trees (Horton 1987). Prescribed fire treatments in pinyon-juniper woodlands could affect Virginia's Warblers in a similar manner. Abundance of Black-throated Gray Warblers decreased after mechanical chaining was used to reduce tree densities in pinyon-juniper woodlands (Sedgwick 1987). Treatments, including prescribed fire, that reduce tree densities and other fuels potentially decrease foraging opportunities for some bird species by removing litter and understory forbs.

PONDEROSA PINE FORESTS

Ponderosa pine spans the full extent of the Rocky Mountains, but considerable variation in stand structure and dynamics occurs across latitudes and elevations (Peet 2000, Schoennagel et al. 2004). Xeric ponderosa pine woodlands dominate montane forests of the southern Rocky Mountains and the lower montane zone of the central and northern Rocky Mountains (Peet 1981). Stand density is relatively low but is often higher in mesic areas with finely textured soils (Peet 1981, Arno 2000). In the upper montane zone and at more northern latitudes, mixed ponderosa pine and Douglas-fir (*Pseudotsuga menziesii*) forests are dominant; we treat these associations as mixed-coniferous forests (Schoennagel et al. 2004). Associated species include aspen (*Populus tremuloides*) in more mesic areas and limber pine (*Pinus flexilis*) along rocky outcrops (Daubenmire 1943, Peet 1981).

Frequent surfaces fires are characteristic of dry, warm woodlands and open-canopy forests, including low-elevation ponderosa pine (Schoennagel et al. 2004). Abundant grasses and forbs contribute to fire initiation and spread, allowing frequent fires. Crown fires are usually rare and small. Short fire intervals, generally 1–50 yr, help to maintain the open structure by killing understory trees and small patches of mature trees (Allen et al. 2002). Fire intervals tend to be shorter in southwestern ponderosa pine than along the Colorado Front Range and Black Hills of Wyoming (Shinneman and Baker 1997, Brown and Sieg 1999, Veblen et al. 2000, Ehle and Baker 2003).

Fire frequency tends to decrease, and severity increase, with increasing altitude and latitude (Veblen et al. 2000, Brown 2004). The most comprehensive fire histories in ponderosa pine are from the American Southwest and southern Rocky Mountains where prior to Euro-American settlement, frequent surface fires predominated (but see Baker and Ehle 2001 for alternative interpretation) and mean fire intervals were short (e.g., 4–36 yr; Swetnam and Baisan 1996). Much longer fire-free periods also have been observed (e.g., 76 yr; Swetnam and Baisan 1996). Longer mean fire-return intervals and fire-free periods are frequently reported in the central and northern Rocky Mountains (Arno et al. 1995, Brown and Sieg 1996, Shinneman and Baker 1997), although stands at grassland ecotones and at lower elevations typically burn more frequently (Barrett and Arno 1982, Brown and Sieg 1999, Veblen et al. 2000).

The historical fire regime in dry, low-elevation ponderosa pine forests has been altered substantially as a result of fire suppression, livestock grazing, and logging and their effects on historical fuel structure (Arno and Gruell 1983, Covington and Moore 1994, Swetnam and Baisan 1996, Veblen et al. 2000, Schoennagel et al. 2004). With reductions in grass fuel, fire intervals have lengthened, and dense stands have developed in which fine fuels are less abundant and ladder fuels carry fire to the canopy (Allen et al. 1998, Schoennagel et al. 2004). Consequently, high-severity fires can strike dry ponderosa pine forests, where historically they were rare. This pattern is well documented for ponderosa pine forests throughout the Rocky Mountain region, including Arizona and New Mexico (e.g., Allen et al. 1998, Moore et al. 1999), some sites in Colorado (e.g., Veblen and Lorenz 1991, Brown et al. 1999, Kaufmann et al. 2000), and portions of Montana (Gruell 1983, Arno et al. 1995).

Evidence of natural, mixed-severity fire regimes is found in some ponderosa pine forests (Mast et al. 1999, Kaufmann et al. 2000, Ehle and Baker 2003). Both surface and crown fires occurred historically in pure or nearly pure ponderosa pine forests of Montana (Arno and Petersen 1983, Arno et al. 1995), the Black Hills of South Dakota (Brown and Sieg

1996, Shinneman and Baker 1997, Brown 2004), and other locations in the Rocky Mountains (e.g., Gruell 1983, Mast et al. 1999, Brown et al. 1999, Ehle and Baker 2003). The relative importance of surface versus crown fires and the size of postfire patches in configuring forests of mixed-severity fire regimes remain uncertain and have likely varied spatially and temporally (Schoennagel et al. 2004).

BIRDS OF PONDEROSA PINE FORESTS

Over 100 bird species use ponderosa pine forests for some portion of their life history (Diem and Zeveloff 1980). Some characteristic species include Flammulated Owl (*Otus flammeolus*), Lewis's Woodpecker, White-headed Woodpecker (*Picoides albolarvatus*), Pygmy Nuthatch (*Sitta pygmaea*), Western Bluebird, and Cassin's Finch (*Carpodacus cassinii*). Partners in Flight priority bird species for ponderosa pine forests of the Rocky Mountains include Flammulated Owl, Lewis's Woodpecker, White-headed Woodpecker, Pygmy Nuthatch, and Cassin's Finch (Partners in Flight 2004). These species require large trees and snags or open canopy provided by this habitat.

BIRD RESPONSE TO FIRE IN PONDEROSA PINE FORESTS

Although avian responses to burned ponderosa pine forests have been studied in the southwestern U.S. (Bock and Block, *this volume*), no studies have examined the effect of fire on avian reproductive success by directly comparing burned and unburned ponderosa pine forests in the Rocky Mountains. To overcome the lack of controlled comparisons, we found reproductive success data for six cavity-nesting species studied in burned ponderosa pine forests in Idaho (2–5 yr postfire; Table 1): Lewis's, Hairy (*Picoides villosus*), Black-backed, and White-headed Woodpeckers, Northern Flicker (*Colaptes auratus*), and Western Bluebird. We then searched the literature for data on the same species nesting in natural cavities in unburned coniferous forests of the West, for comparison. Although many uncontrolled variables occur among these studies, we present the following summary as an exploratory effort in describing patterns of cavity-nesting bird response to fire in ponderosa pine forests.

The nest success values cited in Table 1 were calculated with the Mayfield method (Mayfield 1961) except where we note that apparent nest success was used. The method of apparent nest success contains a known positive bias (Jehle et al. 2004).

Overall, nest success appeared higher for the six species in burned habitats (median nest success = 81.5%, range 70–100%) than in unburned habitats (median = 69%, range 29–100%). Nest success was higher in burned than unburned habitats in 11 of the 14 possible species-by-species comparisons in Table 1, although in two of these 11 the differences were small (< 3%).

We found three interesting exceptions to the general trend of higher nest success in unburned forests. First, Hairy Woodpeckers and Northern Flickers in unburned mixed coniferous-aspen of the Mogollon Rim, Arizona, had essentially the same or greater nest success as individuals in burned ponderosa pine of Idaho (Table 1). The same species nesting in unburned ponderosa pine of Idaho had lower nest success by >20%. In Arizona, these two species nested extensively in aspen (Martin and Li 1992). Many cavity excavators select aspen trees at remarkably high rates compared to their availability (Hutto 1995, Martin et al. 2004); perhaps this tendency is related to high nest success in aspen.

Second, White-headed Woodpeckers had consistently high nest success (>80%) in both burned and unburned ponderosa pine forests of Idaho and Oregon. This species frequently nests in large dead trees but forages in live trees for pine seeds (Dixon 1995, Garrett et al. 1996). White-headed Woodpeckers may benefit from the mosaic of live and dead trees created by low and mixed severity fires.

Third, Western Bluebirds nesting in thinned (i.e., partial tree harvest) or prescribe-burned plots in ponderosa pine forests of Arizona nested with slightly higher success than in the stand-replacement-burned forests in Idaho (75% vs. 70%, respectively). Bluebirds nesting in unburned, untreated ponderosa pine in Arizona had success rates nearly half that recorded in burned ponderosa pine of Idaho (39% vs. 70%, respectively, Table 1). Most nest failures in the Arizona study were due to predation, and fewer potential nest predators were observed in the treated forests (Germaine and Germaine 2002). This comparison gives tentative evidence that prescribed burning and stand-replacement burns in ponderosa pine may result in similar conditions for Western Bluebirds.

A final observation from the nest success values in Table 1 concerns the relative effects of two disturbance types. Black-backed Woodpeckers in burned ponderosa pine had higher nest success than in unburned mixed coniferous forest undergoing a mountain pine beetle (*Dendroctonus ponderosae*) outbreak (87% vs. 69%, respectively, Table 1). This beetle outbreak killed most of the lodgepole pines on the study area and presumably resulted in

TABLE 1. SUMMARY OF AVAILABLE LITERATURE ON THE RESPONSES OF BREEDING BIRDS (CHANGE IN ABUNDANCE) TO WILDFIRE IN FORESTS OF THE ROCKY MOUNTAINS.

Species[a]	State	Year after fire	Size of fires (ha)	No. replicate sites[b]	Response[c]	Habitat/Reference[e]	Comments
Red-tailed Hawk	MT	1–2	25–277,880	33 b	U	Mixed coniferous[1]	
(*Buteo jamaicensis*)	MT	1–4	120, 480	2 b, 3 u	0	Mixed coniferous[2]	Density not estimated; observed in one burned site.
Osprey	MT	1–2	25–277,880	33 b	U	Mixed coniferous[1]	
(*Pandion haliaetus*)	WY	2–5	235, 648	2 b, 2 u	0	Lodgepole and spruce-fir[3]	In burned/unburned forest edge at one site.
American Kestrel	MT	1–2	25–277,880	33 b	U	Mixed coniferous[1]	
(*Falco sparverius*)	MT	1–4	120, 480	2 b, 3 u	+	Mixed coniferous[2]	
	MT	2–6	15,000	2 b, 1 u[g]	+	Lodgepole[4]	Both nest and bird abundance.
Blue Grouse	MT	1–2	25–277,880	33 b	U	Mixed coniferous[1]	
(*Dendragapus obscurus*)	WY	2–5	235, 648	2 b, 2 u	0[d]	Lodgepole and spruce-fir[3]	
	WY	5–10	6, 83	2 b, 6 u	0[d]	Lodgepole[5]	In both burned and unburned sites.
Ruffed Grouse	MT	1–2	25–277,880	33 b	U	Mixed coniferous[1]	
(*Bonasa umbellus*)	MT	1–4	120, 480	2 b, 3 u	0[d]	Mixed coniferous[2]	In both burned and unburned sites.
	WY	2–5	235, 648	2 b, 2 u	–	Lodgepole and spruce-fir[3]	
	WY	1–29[f]	40–1,414	6 b, 6 u	–[d]	Lodgepole and spruce-fir[6]	
Mourning Dove	MT	1–2	25–277,880	33 b	U	Mixed coniferous[1]	
(*Zenaida macroura*)	MT	1–4	120, 480	2 b, 3 u	+	Mixed coniferous[2]	
	WY	5–10	6, 83	2 b, 6 u	0	Lodgepole[5]	In both burned and unburned clearcut forest.
	CO	0–8	43–7,337	8 b, 8 u	+	Mixed coniferous[7]	
Common Nighthawk	MT	1–2	25–277,880	33 b	U	Mixed coniferous[1]	
(*Chordeiles minor*)	MT	1–4	120, 480	2 b, 3 u	+[d]	Mixed coniferous[2]	
	WY	5–10	6, 83	2 b, 6 u	+	Lodgepole[5]	1 yr postfire.
	WY	1–29[f]	40–1,414	6 b, 6 u	0[d]	Lodgepole and spruce-fir[6]	In 7 and 25 yr old burns.
	CO	0–8	43–7,337	8 b, 8 u	+	Mixed coniferous[7]	
Calliope Hummingbird	MT	1–2	25–277,880	33 b	U	Mixed coniferous[1]	
(*Stellula calliope*)	MT	1–4	120, 480	2 b, 3 u	+[d]	Mixed coniferous[2]	
Rufous Hummingbird	MT	1–2	25–277,880	33 b	U	Mixed coniferous[1]	
(*Selasphorus rufus*)	WY	2–5	235, 648	2 b, 2 u	0[d]	Lodgepole and spruce-fir[3]	In burned/unburned forest edge at one site.
Lewis's Woodpecker	MT	1–4	120, 480	2 b, 3 u	0	Mixed coniferous[2]	In one burned site.
(*Melanerpes lewis*)	ID, CO	1–5	89,159; 12,467	2 b, 2 u	+	Ponderosa pine and cottonwood[8]	Comparison of reproductive success in burned coniferous vs. unburned cottonwood forests.
Williamson's Sapsucker	MT	1–2	25–277,880	33 b	U	Mixed coniferous[1]	
(*Sphyrapicus thyroideus*)	MT	1–4	120, 480	2 b, 3 u	0	Mixed coniferous[2]	In burned/unburned forest edge at one site.
	WY	2–5	235, 648	2 b, 2 u	–	Lodgepole and spruce-fir[3]	1 yr postfire.
	WY	1–29[f]	40–1,414	6 b, 6 u	0	Lodgepole and spruce-fir[6]	High densities recorded in 13 yr old burn.
	CO	0–8	43–7,337	8 b, 8 u	–	Mixed coniferous[7]	

TABLE 1. CONTINUED.

Species[a]	State	Year after fire	Size of fires (ha)	No. replicate sites[b]	Response[c]	Habitat/Reference[e]	Comments
Red-naped Sapsucker	MT	1–2	25–277,880	33 b	U	Mixed coniferous[1]	
(*Sphyrapicus nuchalis*)	MT	1–4	120, 480	2 b, 3 u	0	Mixed coniferous[2]	Reported as Yellow-bellied Sapsucker (*Sphyrapicus varius*); in burned/unburned forest edge at one site.
	MT	2–6	15,000	2 b, 1 u[g]	–	Lodgepole[4]	Both nest and bird abundance.
	WY	1–29[f]	40–1,414	6 b, 6 u	0[d]	Lodgepole and spruce-fir[6]	Reported as Yellow-bellied Sapsucker (*Sphyrapicus varius*); in 43 yr old burn.
	CO	0–8	43–7,337	8 b, 8 u	+	Mixed coniferous[7]	
Downy Woodpecker	MT	1–2	25–277,880	33 b	U	Mixed coniferous[1]	
(*Picoides pubescens*)	MT	1–4	120, 480	2 b, 3 u	0[d]	Mixed coniferous[2]	
	MT	2–6	15,000	2 b, 1 u[g]	+	Lodgepole[4]	Both nest and bird abundance.
Hairy Woodpecker	MT	1–2	25–277,880	33 b	C	Mixed coniferous[1]	
(*Picoides villosus*)	MT	1–4	120, 480	2 b, 3 u	m	Mixed coniferous[2]	
	WY	2–5	235, 648	2 b, 2 u	m	Lodgepole and spruce-fir[3]	
	MT	2–6	15,000	2 b, 1 u[g]	+	Lodgepole[4]	Both nest and bird abundance.
	WY	5–10	6, 83	2 b, 6 u	+	Lodgepole[5]	
	WY	1–29[f]	40–1,414	6 b, 6 u	+	Lodgepole and spruce-fir[6]	
	CO	0–8	43–7,337	8 b, 8 u	+	Mixed coniferous[7]	
	WY, ID	1–2	3400[h]	1 b, 2 u	+	Lodgepole[9]	Highest nest densities in burned forests.
Three-toed Woodpecker	MT	1–2	25–277,880	33 b	F	Mixed coniferous[1]	
(*Picoides tridactylus*)	MT	1–4	120, 480	2 b, 3 u	+	Mixed coniferous[2]	
	WY	2–5	235, 648	2 b, 2 u	+	Lodgepole and spruce-fir[3]	
	MT	2–6	15,000	2 b, 1 u[g]	+ and 0	Lodgepole[4]	Positive for nest abundance; no significant difference observed in bird abundance.
	WY	1–29[f]	40–1,414	6 b, 6 u	+	Lodgepole and spruce-fir[6]	
	CO	0–8	43–7,337	8 b, 8 u	+	Mixed coniferous[7]	
	WY, ID	1–2	3,400[h]	1 b, 2 u	+	Lodgepole[9]	
Black-backed Woodpecker	MT	1–2	25–277,880	33 b	F	Mixed coniferous[1]	
(*Picoides arcticus*)	MT	1–4	120, 480	2 b, 3 u	+	Mixed coniferous[2]	
	WY	2–5	235, 648	2 b, 2 u	+	Lodgepole and spruce-fir[3]	1 yr postfire.
	MT	2–6	15,000	2 b, 1 u[g]	+	Lodgepole[4]	Both nest and bird abundance.
	WY	5–10	6, 83	2 b, 6 u	–	Lodgepole[5]	1 yr postfire.
	WY	1–29[f]	40–1,414	6 b, 6 u	+	Lodgepole and spruce-fir[6]	
	WY, ID	1–2	3,400[h]	1 b, 2 u	+	Lodgepole[9]	

TABLE 1. CONTINUED.

Species[a]	State	Year after fire	Size of fires (ha)	No. replicate sites[b]	Response[c]	Habitat/Reference[e]	Comments
Northern Flicker	MT	1–2	25–277,880	33 b	C	Mixed coniferous[1]	
(*Colaptes auratus*)	WY	2–5	235, 648	2 b, 2 u	+	Lodgepole and spruce-fir[3]	
	MT	2–6	15,000	2 b, 1 u[g]	+ and 0	Lodgepole[4]	Positive for nest abundance; no significant difference observed in bird abundance.
	WY	5–10	6, 83	2 b, 6 u	+	Lodgepole[5]	Densities highest 1 yr postfire.
	WY	1–29[f]	40–1,414	6 b, 6 u	+	Lodgepole and spruce-fir[6]	
	CO	0–8	43–7,337	8 b, 8 u	m	Mixed coniferous[7]	
Pileated Woodpecker	MT	1–2	25–277,880	33 b	U	Mixed coniferous[1]	
(*Dryocopus pileatus*)	MT	1–4	120, 480	2 b, 3 u	0	Mixed coniferous[2]	In burned/unburned forest edge at one site.
	MT	2–6	15,000	2 b, 1 u[g]	–	Lodgepole[4]	
Olive-sided Flycatcher	MT	1–2	25–277,880	33 b	F	Mixed coniferous[1]	
(*Contopus cooperi*)	MT	1–4	120, 480	2 b, 3 u	0	Mixed coniferous[2]	
	WY	2–5	235, 648	2 b, 2 u	0	Lodgepole and spruce-fir[3]	In both burned forest and unburned forest edge.
	WY	5–10	6, 83	2 b, 6 u	m	Lodgepole[5]	
	CO	0–8	43–7,337	8 b, 8 u	+	Mixed coniferous[7]	
Western Wood-Pewee	MT	1–2	25–277,880	33 b	F	Mixed coniferous[1]	
(*Contopus sordidulus*)	MT	1–4	120, 480	2 b, 3 u	0	Mixed coniferous[2]	High densities at one site 4 yr after fire.
	WY	2–5	235, 648	2 b, 2 u	+	Lodgepole and spruce-fir[3]	
	WY	5–10	6, 83	2 b, 6 u	+	Lodgepole[5]	
	WY	1–29[f]	40–1,414	6 b, 6 u	+	Lodgepole and spruce-fir[6]	
	CO	0–8	43–7,337	8 b, 8 u	+	Mixed coniferous[7]	
Hammond's Flycatcher	MT	1–2	25–277,880	33 b	F	Mixed coniferous[1]	
(*Empidonax hammondii*)	MT	1–4	120, 480	2 b, 3 u	m	Mixed coniferous[2]	
	CO	0–8	43–7,337	8 b, 8 u	m	Mixed coniferous[7]	
Dusky Flycatcher (*Empidonax oberholseri*)	MT	1–2	25–277,880	33 b	F	Mixed coniferous[1]	
Warbling Vireo	MT	1–2	25–277,880	33 b	F	Mixed coniferous[1]	
(*Vireo gilvus*)	MT	1–4	120, 480	2 b, 3 u	0	Mixed coniferous[2]	
	CO	0–8	43–7,337	8 b, 8 u	–	Mixed coniferous[7]	
Plumbeous/Cassin's Vireo (*V. plumbeus/V. cassinii*)	MT	1–2	25–277,880	33 b	U	Mixed coniferous[1]	Reported as Solitary Vireo (*V. solitarius*).
	MT	1–4	120, 480	2 b, 3 u	0	Mixed coniferous[2]	Present as Solitary Vireo (*V. solitarius*); in burned/unburned forest edge at one site.
	CO	0–8	43–7,337	8 b, 8 u	–	Mixed coniferous[7]	

Table 1. Continued.

Species[a]	State	Year after fire	Size of fires (ha)	No. replicate sites[b]	Response[c]	Habitat/Reference[e]	Comments
Gray Jay	MT	1–2	25–277,880	33 b	U	Mixed coniferous[1]	
(*Perisoreus canadensis*)	WY	2–5	235, 648	2 b, 2 u	–	Lodgepole and spruce-fir[3]	
	WY	5–10	6, 83	2 b, 6 u	m	Lodgepole[5]	
	WY	1–29[f]	40–1,414	6 b, 6 u	m	Lodgepole and spruce-fir[6]	
	CO	0–8	43–7,337	8 b, 8 u	m	Mixed coniferous[7]	
Steller's Jay	MT	1–2	25–277,880	33 b	U	Mixed coniferous[1]	
(*Cyanocitta stelleri*)	MT	1–4	120, 480	2 b, 3 u	0	Mixed coniferous[2]	In one burned site.
	CO	0–8	43–7,337	8 b, 8 u	–	Mixed coniferous[7]	
Clark's Nutcracker	MT	1–2	25–277,880	33 b	F	Mixed coniferous[1]	
(*Nucifraga columbiana*)	MT	1–4	120, 480	2 b, 3 u	m	Mixed coniferous[2]	
	WY	2–5	235, 648	2 b, 2 u	–	Lodgepole and spruce-fir[3]	
	WY	5–10	6, 83	2 b, 6 u	0	Lodgepole[5]	Low numbers at both burned and unburned sites.
	WY	1–29[f]	40–1,414	6 b, 6 u	+	Lodgepole and spruce-fir[6]	>4 yr postfire.
	CO	0–8	43–7,337	8 b, 8 u	+	Mixed coniferous[7]	
Common Raven	MT	1–2	25–77,880	33 b	F	Mixed coniferous[1]	
(*Corvus corax*)	MT	1–4	120, 480	2 b, 3 u	m	Mixed coniferous[2]	
	WY	2–5	235, 648	2 b, 2 u	0	Lodgepole and spruce-fir[3]	Low numbers in burned and burned/unburned forest edge.
Tree Swallow	MT	1–2	25–277,880	33 b	U	Mixed coniferous[1]	
(*Tachycineta bicolor*)	WY	2–5	235, 648	2 b, 2 u	+	Lodgepole and spruce-fir[3]	
	MT	2–6	15,000	2 b, 1 u[g]	+	Lodgepole[4]	For both nest and bird abundance.
	WY	5–10	6, 83	2 b, 6 u	0	Lodgepole[5]	
	WY	1–29[f]	40–1,414	6 b, 6 u	+	Lodgepole and spruce-fir[6]	>4 yr postfire.
	CO	0–8	43–7,337	8 b, 8 u	+	Mixed coniferous[7]	
Black-capped Chickadee	MT	1–2	25–277,880	33 b	F	Mixed coniferous[1]	
(*Poecile atricapilla*)	WY	2–5	235, 648	2 b, 2 u	0	Lodgepole and spruce-fir[3]	Low numbers in burned/unburned forest edge.
Mountain Chickadee	MT	1–2	25–277,880	33 b	F	Mixed coniferous[1]	
(*Poecile gambeli*)	WY	2–5	235, 648	2 b, 2 u	–	Lodgepole and spruce-fir[3]	
	WY	1–29[f]	40–1,414	6 b, 6 u	–	Lodgepole and spruce-fir[6]	
	CO	0–8	43–7,337	8 b, 8 u	–	Mixed coniferous[7]	
Chickadee (*Poecile* spp.)	MT	2–6	15,000	2 b, 1 u[g]	–	Lodgepole[4]	Includes *P. atricapilla*, *gambeli*, and *rufescens*; negative for both nest and bird abundance.
Chickadee (*Poecile* spp.)	MT	1–4	120, 480	2 b, 3 u	–	Mixed coniferous[2]	No distinction made between *P. atricapilla* and *gambeli.*

TABLE 1. CONTINUED.

Species[a]	State	Year after fire	Size of fires (ha)	No. replicate sites[b]	Response[c]	Habitat/Reference[e]	Comments
Red-breasted Nuthatch	MT	1–2	25–277,880	33 b	F	Mixed coniferous[1]	
(*Sitta canadensis*)	MT	1–4	120, 480	2 b, 3 u	–	Mixed coniferous[2]	
	WY	2–5	235, 648	2 b, 2 u	0	Lodgepole and spruce-fir[3]	Low numbers on both burned and unburned sites.
	MT	2–6	15,000	2 b, 1 u[g]	–	Lodgepole[4]	For both nest and bird abundance.
	WY	1–29[f]	40–1,414	6 b, 6 u	–	Lodgepole and spruce-fir[6]	
	CO	0–8	43–7,337	8 b, 8 u	m	Mixed coniferous[7]	
Pygmy Nuthatch	CO	0–8	43–7,337	8 b, 8 u	–	Mixed coniferous[7]	
(*Sitta pygmaea*)							
Brown Creeper	MT	1–2	25–277,880	33 b	U	Mixed coniferous[1]	
(*Certhia americana*)	MT	1–4	120, 480	2 b, 3 u	0	Mixed coniferous[2]	Low numbers in unburned forest adjacent to forest burn.
	WY	2–5	235, 648	2 b, 2 u	–	Lodgepole and spruce-fir[3]	
	WY	1–29[f]	40–1,414	6 b, 6 u	m	Lodgepole and spruce-fir[6]	Moderate severity portions of burn.
	CO	0–8	43–7,337	8 b, 8 u	–	Mixed coniferous[7]	
House Wren	MT	1–2	25–277,880	33 b	U	Mixed coniferous[1]	
(*Troglodytes aedon*)	MT	1–4	120, 480	2 b, 3 u	+	Mixed coniferous[2]	
	MT	2–6	15,000	2 b, 1 u[g]	+	Lodgepole[4]	For both nest and bird abundance.
	WY	1–29[f]	40–1,414	6 b, 6 u	+	Lodgepole and spruce-fir[6]	Moderate severity or >6 yr postfire.
	CO	0–8	43–7,337	8 b, 8 u	m	Mixed coniferous[7]	
Rock Wren	MT	1–2	25–277,880	33 b	U	Mixed coniferous[1]	
(*Salpinctes obsoletus*)	CO	0–8	43–7,337	8 b, 8 u	+	Mixed coniferous[7]	
Golden-crowned Kinglet	MT	1–2	25–277,880	33 b	U	Mixed coniferous[1]	
(*Regulus satrapa*)	MT	1–4	120, 480	2 b, 3 u	–	Mixed coniferous[2]	
	WY	5–10	6, 83	2 b, 6 u	0	Lodgepole[5]	In one unburned site.
	WY	1–29[f]	40–1,414	6 b, 6 u	–	Lodgepole and spruce-fir[6]	
	CO	0–8	43–7,337	8 b, 8 u	–	Mixed coniferous[7]	
Ruby-crowned Kinglet	MT	1–2	25–277,880	33 b	F	Mixed coniferous[1]	
(*Regulus calendula*)	MT	1–4	120, 480	2 b, 3 u	–	Mixed coniferous[2]	
	WY	2–5	235, 648	2 b, 2 u	m[d]	Lodgepole and spruce-fir[3]	
	WY	5–10	6, 83	2 b, 6 u	0	Lodgepole[5]	Low numbers in one burned and one unburned site.
	WY	1–29[f]	40–1,414	6 b, 6 u	–	Lodgepole and spruce-fir[6]	
	CO	0–8	43–7,337	8 b, 8 u	m	Mixed coniferous[7]	
Swainson's Thrush	MT	1–2	25–277,880	33 b	F	Mixed coniferous[1]	
(*Catharus ustulatus*)	MT	1–4	120, 480	2 b, 3 u	0	Mixed coniferous[2]	
	WY	1–29[f]	40–1,414	6 b, 6 u	m	Lodgepole and spruce-fir[6]	

Table 1. Continued.

Species[a]	State	Year after fire	Size of fires (ha)	No. replicate sites[b]	Response[c]	Habitat/Reference[e]	Comments
Hermit Thrush	MT	1–2	25–277,880	33 b	F	Mixed coniferous[1]	
(*Catharus guttatus*)	WY	2–5	235, 648	2 b, 2 u	–	Lodgepole and spruce-fir[3]	
	WY	5–10	6, 83	2 b, 6 u	m	Lodgepole[5]	
	WY	1–29[f]	40–1,414	6 b, 6 u	–	Lodgepole and spruce-fir[6]	
	CO	0–8	43–7,337	8 b, 8 u	–	Mixed coniferous[7]	
American Robin	MT	1–2	25–277,880	33 b	C	Mixed coniferous[1]	
(*Turdus migratorius*)	MT	1–4	120, 480	2 b, 3 u	m	Mixed coniferous[2]	
	WY	2–5	235, 648	2 b, 2 u	m	Lodgepole and spruce-fir[3]	
	WY	5–10	6, 83	2 b, 6 u	m	Lodgepole[5]	
	WY	1–29[f]	40–1,414	6 b, 6 u	+	Lodgepole and spruce-fir[6]	
	CO	0–8	43–7,337	8 b, 8 u	m	Mixed coniferous[7]	
Varied Thrush	MT	1–2	25–277,880	33 b	U	Mixed coniferous[1]	
(*Ixoreus naevius*)	MT	1–4	120, 480	2 b, 3 u	0	Mixed coniferous[2]	
Townsend's Solitaire	MT	1–2	25–277,880	33 b	C	Mixed coniferous[1]	
(*Myadestes townsendi*)	MT	1–4	120, 480	2 b, 3 u	m	Mixed coniferous[2]	
	WY	5–10	6, 83	2 b, 6 u	m	Lodgepole[5]	
	WY	1–29[f]	40–1,414	6 b, 6 u	m	Lodgepole and spruce-fir[6]	In 7 yr old burn.
	CO	0–8	43–7,337	8 b, 8 u	m	Mixed coniferous[7]	
Mountain Bluebird	MT	1–2	25–277,880	33 b	F	Mixed coniferous[1]	
(*Siala currucoides*)	MT	1–4	120, 480	2 b, 3 u	+	Mixed coniferous[2]	
	WY	2–5	235, 648	2 b, 2 u	+	Lodgepole and spruce-fir[3]	
	MT	2–6	15,000	2 b, 1 u[g]	+	Lodgepole[4]	For both nest and bird abundance.
	WY	5–10	6, 83	2 b, 6 u	+	Lodgepole[5]	
	WY	1–29[f]	40–1,414	6 b, 6 u	+	Lodgepole and spruce-fir[6]	
	CO	0–8	43–7,337	8 b, 8 u	+	Mixed coniferous[7]	
Western Bluebird	MT	1–2	25–277,880	33 b	U	Mixed coniferous[1]	
(*Siala mexicana*)	CO	0–8	43–7,337	8 b, 8 u	+	Mixed coniferous[7]	
European Starling	MT	1–2	25–277,880	33 b	U	Mixed coniferous[1]	
(*Sturnus vulgaris*)	MT	1–4	120, 480	2 b, 3 u	0	Mixed coniferous[2]	
Yellow-rumped Warbler	MT	1–2	25–277,880	33 b	C	Mixed coniferous[1]	
(*Dendroica coronata*)	MT	1–4	120, 480	2 b, 3 u	m	Mixed coniferous[2]	
	WY	2–5	235, 648	2 b, 2 u	m	Lodgepole and spruce-fir[3]	
	WY	5–10	6, 83	2 b, 6 u	+	Lodgepole[5]	
	WY	1–29[f]	40–1,414	6 b, 6 u	–	Lodgepole and spruce-fir[6]	
	CO	0–8	43–7,337	8 b, 8 u	–	Mixed coniferous[7]	

TABLE 1. CONTINUED.

Species[a]	State	Year after fire	Size of fires (ha)	No. replicate sites[b]	Response[c]	Habitat/Reference[e]	Comments
Townsend's Warbler	MT	1–2	25–277,880	33 b	F	Mixed coniferous[1]	
(*Dendroica townsendi*)	MT	1–4	120, 480	2 b, 3 u	0	Mixed coniferous[2]	
	WY	2–5	235, 648	2 b, 2 u	0	Lodgepole and spruce-fir[3]	In one burned site.
Orange-crowned Warbler	MT	1–2	25–277,880	33 b	U	Mixed coniferous[1]	
(*Vermivora celata*)	MT	1–4	120, 480	2 b, 3 u	0	Mixed coniferous[2]	
MacGillivray's Warbler	MT	1–2	25–277,880	33 b	F	Mixed coniferous[1]	
(*Oporornis tolmiei*)	MT	1–4	120, 480	2 b, 3 u	m	Mixed coniferous[2]	
Wilson's Warbler	MT	1–2	25–277,880	33 b	U	Mixed coniferous[1]	
(*Wilsonia pusilla*)	WY	2–5	235, 648	2 b, 2 u	0	Lodgepole and spruce-fir[3]	In burned/unburned forest edge.
Western Tanager	MT	1–2	25–277,880	33 b	C	Mixed coniferous[1]	
(*Piranga ludoviciana*)	MT	1–4	120, 480	2 b, 3 u	m	Mixed coniferous[2]	
	WY	2–5	235, 648	2 b, 2 u	m[d]	Lodgepole and spruce-fir[3]	
	WY	5–10	6, 83	2 b, 6 u	+	Lodgepole[5]	
	WY	1–29[f]	40–1,414	6 b, 6 u	–	Lodgepole and spruce-fir[6]	
	CO	0–8	43–7,337	8 b, 8 u	m	Mixed coniferous[7]	
Lazuli Bunting	MT	1–2	25–277,880	33 b	F	Mixed coniferous[1]	
(*Passerina amoena*)	MT	1–4	120, 480	2 b, 3 u	0	Mixed coniferous[2]	
Chipping Sparrow	MT	1–2	25–277,880	33 b	C	Mixed coniferous[1]	
(*Spizella passerina*)	MT	1–4	120, 480	2 b, 3 u	m	Mixed coniferous[2]	
	WY	2–5	235, 648	2 b, 2 u	m	Lodgepole and spruce-fir[3]	
	WY	5–10	6, 83	2 b, 6 u	+	Lodgepole[5]	
	WY	1–29[f]	40–1,414	6 b, 6 u	m	Lodgepole and spruce-fir[6]	
	CO	0–8	43–7,337	8 b, 8 u	m	Mixed coniferous[7]	
Fox Sparrow	MT	1–2	25–277,880	33 b	U	Mixed coniferous[1]	
(*Passerella iliaca*)	WY	5–10	6, 83	2 b, 6 u	+	Lodgepole[5]	
Song Sparrow	MT	1–2	25–277,880	33 b	U	Mixed coniferous[1]	
(*Melospiza melodia*)	WY	1–29[f]	40–1,414	6 b, 6 u	0	Lodgepole and spruce-fir[6]	
Lincoln's Sparrow	MT	1–2	25–277,880	33 b	F	Mixed coniferous[1]	
(*Melospiza lincolnii*)	WY	2–5	235, 648	2 b, 2 u	+	Lodgepole and spruce-fir[3]	
White-crowned Sparrow	MT	1–2	25–277,880	33 b	U	Mixed coniferous[1]	
(*Zonotrichia leucophrys*)	WY	2–5	235, 648	2 b, 2 u	+	Lodgepole and spruce-fir[3]	
	WY	5–10	6, 83	2 b, 6 u	0	Lodgepole[5]	
	CO	0–8	43–7,337	8 b, 8 u	+	Mixed coniferous[7]	

TABLE 1. CONTINUED.

Species[a]	State	Year after fire	Size of fires (ha)	No. replicate sites[b]	Response[c]	Habitat/Reference[e]	Comments
Dark-eyed Junco	MT	1–2	25–277,880	33 b	C	Mixed coniferous[1]	
(*Junco hyemalis*)	MT	1–4	120, 480	2 b, 3 u	m	Mixed coniferous[2]	
	WY	2–5	235, 648	2 b, 2 u	+	Lodgepole and spruce-fir[3]	Highest densities in burned forest but present in high densities in unburned forest.
	WY	5–10	6, 83	2 b, 6 u	+	Lodgepole[5]	Listed as Gray-headed Junco (*Junco caniceps*); highest densities in burned forest but present in high densities in unburned forest.
	WY	1–29[f]	40–1,414	6 b, 6 u	m	Lodgepole and spruce-fir[6]	Listed as Oregon Junco (*J. hyemalis oreganos*).
	CO	0–8	43–7,337	8 b, 8 u	m	Mixed coniferous[7]	
Brown-headed Cowbird	MT	1–2	25–277,880	33 b	F	Mixed coniferous[1]	
(*Molothrus ater*)	WY	2–5	235, 648	2 b, 2 u	0	Lodgepole and spruce-fir[3]	
Brewer's Blackbird	MT	1–2	25–277,880	33 b	U	Mixed coniferous[1]	
(*Euphagus cyanocephalus*)	MT	1–4	120, 480	2 b, 3 u	0	Mixed coniferous[2]	
Evening Grosbeak	MT	1–2	25–277,880	33 b	U	Mixed coniferous[1]	
(*Coccothraustes vespertinus*)	MT	1–4	120, 480	2 b, 3 u	–	Mixed coniferous[2]	
	CO	0–8	43–7,337	8 b, 8 u	m	Mixed coniferous[7]	
Pine Grosbeak	MT	1–2	25–277,880	33 b	U	Mixed coniferous[1]	
(*Pinicola enucleator*)	WY	2–5	235, 648	2 b, 2 u	–	Lodgepole and spruce-fir[3]	
	WY	5–10	6, 83	2 b, 6 u	0	Lodgepole5	
	WY	1–29[f]	40–1,414	6 b, 6 u	–[d]	Lodgepole and spruce-fir[6]	
Cassin's Finch	MT	1–2	25–277,880	33 b	F	Mixed coniferous[1]	
(*Carpodacus cassinii*)	MT	1–4	120, 480	2 b, 3 u	m	Mixed coniferous[2]	
	WY	2–5	235, 648	2 b, 2 u	+	Lodgepole and spruce-fir[3]	
	WY	5–10	6, 83	2 b, 6 u	+	Lodgepole[5]	
	WY	1–29[f]	40–1,414	6 b, 6 u	+	Lodgepole and spruce-fir[6]	>2 yr postfire.
	CO	0–8	43–7,337	8 b, 8 u	+	Mixed coniferous[7]	
Red Crossbill	MT	1–2	25–277,880	33 b	F	Mixed coniferous[1]	
(*Loxia curvirostra*)	MT	1–4	120, 480	2 b, 3 u	–	Mixed coniferous[2]	
	WY	2–5	235, 648	2 b, 2 u	0[d]	Lodgepole and spruce-fir[3]	
	WY	1–29[f]	40–1,414	6 b, 6 u	0[d]	Lodgepole and spruce-fir[6]	
	CO	0–8	43–7,337	8 b, 8 u	m	Mixed coniferous[7]	
Pine Siskin	MT	1–2	25–277,880	33 b	C	Mixed coniferous[1]	
(*Carduelis pinus*)	MT	1–4	120, 480	2 b, 3 u	m	Mixed coniferous[2]	
	WY	2–5	235, 648	2 b, 2 u	+	Lodgepole and spruce-fir[3]	

TABLE 1. CONTINUED.

Species[a]	State	Year after fire	Size of fires (ha)	No. replicate sites[b]	Response[c]	Habitat/Reference[e]	Comments
Pine Siskin (*continued*)	WY	5–10	6, 83	2 b, 6 u	m	Lodgepole[5]	
(*Carduelis pinus*)	WY	1–29[f]	40–1,414	6 b, 6 u	m[d]	Lodgepole and spruce-fir[6]	
	CO	0–8	43–7,337	8 b, 8 u	m	Mixed coniferous[7]	

[a] Species not included in the table proper were those reported by only one study and recorded as uncommon by Hutto (1995) or as + by other references listed in footnote e, including Northern Goshawk (*Accipiter gentilis*), Ferruginous Hawk (*Buteo regalis*), Band-tailed Pigeon (*Patagioenas fasciata*), Flammulated Owl (*Otus flammeolus*), Black-chinned Hummingbird (*Archilochus alexandri*), White-headed Woodpecker (*Picoides albolarvatus*), Western Scrub-Jay (*Aphelocoma californica*), Pinyon Jay (*Gymnorhinus cyanocephalus*), Gray Flycatcher (*Empidonax wrightii*), Cassin's Kingbird (*Tyrannus vociferans*), Gray Vireo (*Vireo vicinior*), Bushtit (*Psaltriparus minimus*), Juniper Titmouse (*Baeolophus ridgwayi*), Virginia's Warbler (*Vermivora virginiae*), Scott's Oriole (*Icterus parisorum*), Evening Grosbeak (*Coccothraustes vespertinus*).

[b] All studies were of wildfire; b = number of burned sites; u = number of unburned sites.

[c] + = increase; – = decrease; 0 = no effect or study inconclusive; m = mixed response; for responses without a comparison to unburned forests (i.e., Hutto 1995), frequency of occurrence was classified as U = uncommon (< 25% of burns), F = frequently observed (25–75% of burns) and C= common (> 75% of burns).

[d] Low detections, < 0.05 birds/ha.

[e] References: 1 = Hutto 1995; 2 = Harris 1982; 3 = Pfister 1980; 4 = Caton 1996; 5 = Davis 1976; 6 = Taylor and Barmore 1980; 7 = Kotliar et al. 2002; 8 = Saab and Vierling 2001; 9 = Hoffman 1997.

[f] Sites censused > 29 yr after fire were considered unburned.

[g] Number of unburned replicates not reported.

[h] Area surveyed within burned forest; size of fire not reported.

a flush of beetle larvae available as food for Black-backed Woodpeckers (Goggans et al. 1988). Such an increase in woodpecker prey is qualitatively similar to the increase in wood-boring beetle larvae that accompanies stand-replacing fire, inviting the suggestion that fire and bark beetle outbreaks create similar habitat conditions for woodpeckers. However, bark and wood-boring beetles have marked ecological differences that affect their value as woodpecker prey (Mitton and Sturgeon 1982, Powell 2000). Bark beetle outbreaks almost certainly offer more woodpecker prey than unburned forests without outbreaks, but they are not necessarily as abundant in prey as burned forests (Powell 2000).

One study in Table 1 measured a reproductive success variable other than nest success for Lewis's Woodpecker (Saab and Vierling 2001), a species well known to strongly favor burned forests (e.g., Tobalske 1997, Linder and Anderson 1998). Saab and Vierling (2001) compared productivity of Lewis's Woodpeckers between burned ponderosa pine of Idaho and unburned cottonwood (*Populus fremontii*) riparian forests of Colorado. Nests in burned ponderosa pine had nearly double the productivity of nests in unburned cottonwood riparian (0.69 vs. 0.38 female fledglings per female per year, respectively), leading the authors to suggest that burned ponderosa pine forest may be a source habitat for Lewis's Woodpeckers.

The cavity-nesting birds reviewed here breed with relatively high success in stand-replacement burns of ponderosa pine forest. High reproductive success and increased productivity in recently burned forests might be explained in part by a reduction or elimination of nest predators following stand-replacement fires (Saab and Vierling 2001). Fire management of ponderosa pine forests in the Rocky Mountains has emphasized prescribed, understory fire to restore ecosystem function (e.g., Arno 2000). Stand-replacement fire may be equally important in maintaining some ponderosa pine forests (Veblen et al. 2000, Baker and Ehle 2001, Ehle and Baker 2003), and for the long-term persistence of cavity-nesting birds that thrive in these habitats. We found no published studies that investigated the effects of prescribed fire on birds in the southern and central Rocky Mountains. Such studies are needed to understand the ecological consequences of managing forests with prescribed fire, fire exclusion, or wildland fire.

MIXED CONIFEROUS FORESTS

Mixed coniferous (mesic montane) forests occur predominantly at mid-elevations, where the topographic variation creates a mosaic of tree species and densities (Peet 2000). In the central Rocky Mountains, Douglas-fir often occurs with white fir (*Abies concolor*), blue spruce (*Picea pungens*), ponderosa pine, limber pine, and quaking aspen; in the northern Rocky Mountains, Douglas-fir, grand fir (*Abies grandis*), ponderosa pine, and western larch (*Larix occidentalis*) are associated species (Daubenmire 1943).

On the west slope of the northern Rocky Mountains, mesic cedar-hemlock (*Thuja-Tsuga*; Cascadian) forests occur as a result of the Pacific maritime influence (Daubenmire 1943, Peet 2000). Dominant species include western hemlock (*Tsuga heterophylla*), western redcedar (*Thuja plicata*), grand fir, and Pacific yew (*Taxus brevifolia*) (Daubenmire 1943). These forests resemble those found in the western Cascade Mountains (Peet 2000).

Mixed-severity fire regimes are characteristic of mixed coniferous forests (Schoennagel et al. 2004). For example, mixed coniferous forest in western Montana burned in stand-replacement fires at long intervals of 150 to >400 yr, while low severity, understory fires burned at short intervals (20–30 yr averages) (see Arno 2000).

In mixed-severity regimes, the extent of postfire tree mortality varies from sparse to complete, depending on the severity of the surface fire. The variation in fire behavior inherent in mixed-severity regimes results in complex forest age structures within burns (Agee 1998). Upper-montane ponderosa pine forests, especially those with a greater component of Douglas-fir, typically experienced both frequent surface fires and infrequent crown fires (i.e., a mixed-severity regime).

Reductions of fire activity in mixed coniferous forests began in the early twentieth century as a result of livestock grazing (removing fine fuels), fire exclusion, and logging (Arno 2000). The densities of relatively fire intolerant and shade tolerant species, such as Douglas-fir and grand fir, have increased in response (Arno et al. 1995, Kaufmann et al. 2000). This is particularly evident within the mixed coniferous zone at lower elevations, on drier aspects, and adjacent to grasslands where fires historically were more frequent (Schoennagel et al. 2004). In some areas, removal of overstory trees in more than a century of logging has contributed to thickets of relatively small trees (Kaufmann et al. 2000). An increase in forest disturbance (e.g., logging, fires) in many areas of the Rocky Mountains during early Euro-American settlement probably synchronized large areas of the landscape and increased aspen coverage, which subsequently diminished by the late twentieth century in many areas due to senescence and encroachment by conifers (Veblen 2000).

BIRDS OF MIXED CONIFEROUS FORESTS

Sanderson et al. (1980) list 96 species that use mixed coniferous forests. Of 166 bird species detected during point count visits conducted across a variety of habitats in the northern Rocky Mountains, 75 were detected in mixed coniferous forests (Hutto and Young 1999). Some characteristic species include Northern Goshawk (*Accipiter gentilis*), Blue Grouse (*Dendragapus obscurus*), Williamson's Sapsucker (*Sphyrapicus thyroideus*), Hairy Woodpecker, Hammond's Flycatcher (*E. hammondii*), Mountain Chickadee (*Poecile gambeli*), Brown Creeper (*Certhia americana*), Golden-crowned Kinglet (*Regulus satrapa*), Ruby-crowned Kinglet (*R. calendula*), Hermit Thrush (*Catharus guttatus*), Yellow-rumped Warbler (*Dendroica coronata*), Western Tanager (*Piranga ludoviciana*), Evening Grosbeak (*Coccothraustes vespertinus*), and Pine Siskin (*Carduelis pinus*). Partners in Flight priority bird species for mixed coniferous forests of the Rocky Mountains include Northern Goshawk, Williamson's Sapsucker, and Brown Creeper due to their need for high canopy closure and high densities of large diameter trees (Partners in Flight 2004).

BIRD RESPONSE TO FIRE IN MIXED CONIFEROUS FORESTS

Generalizations regarding bird response to fire in mixed coniferous forests are difficult due to variation in topography, stand densities, forest structure, fire history, and tree and understory species composition. Most data available on avian response to fire in mixed coniferous forests come from a handful of studies (Harris 1982, Hutto 1995, Hitchcox 1996, Kotliar et al. 2002).

Of these studies, only Harris (1982) and Kotliar et al. (2002) directly compared bird response in burned and unburned mixed coniferous forests (Table 2). Although Hutto (1995) did not compare abundance between burned and unburned forests, he did report the relative occurrence of 87 species within 33 burned forests. Hutto (1995) and Kotliar et al. (2002) did not distinguish between different types of burned forest, so we include them in this section only. Species responses were based on frequency of occurrence (Hutto 1995), abundance estimates from point counts (Kotliar et al. 2002), and fixed-width transect surveys (Harris 1982). These techniques

TABLE 2. COMPARISON OF REPRODUCTIVE SUCCESS FOR SIX BIRD SPECIES IN BURNED AND UNBURNED HABITATS. NUMBERS ARE MAYFIELD-CORRECTED PERCENT SUCCESS EXCEPT WHERE OTHERWISE INDICATED[a].

	Burned forest			Unburned forest		
Species	N	Nest success (%)	Habitat	N	Nest success (%)	Habitat
Lewis's Woodpecker	283	78	Ponderosa pine, Idaho[b]	65	46	Cottonwood riparian, Colorado[b]
		0.69[c]			0.38[c]	
Hairy Woodpecker	91	75	Ponderosa pine, Idaho[d]	8	74	Mixed coniferous-aspen, Arizona[e]
				29	48	Ponderosa pine, Idaho[f]
Black-backed Woodpecker	33	87	Ponderosa pine, Idaho[d]	19	69[a]	Mixed coniferous, beetle-killed, Oregon[g]
	14	100[a]	Lodgepole pine, Wyoming[h]			
White-headed Woodpecker	6	100	Ponderosa pine, Idaho[d]	41	83[a]	Ponderosa pine, Oregon[i]
	20	85	Ponderosa pine, Oregon[f]	16	88	Ponderosa pine, Oregon[i]
Northern Flicker	97	70	Ponderosa pine, Idaho[d]	34	100	Mixed coniferous-aspen, Arizona[e]
				48	49	Ponderosa pine, Idaho[f]
Western Bluebird	100	70	Ponderosa pine, Idaho[d]	41	39	Ponderosa pine, untreated, Arizona[j]
				56	75	Ponderosa pine, thinned or prescribe-burned, Arizona[j]
				39	29	Oak-pine, California[k]

[a] Values are apparent nest success.
[b] Saab and Vierling 2001.
[c] Variable is productivity, in number of female fledglings per female per year.
[d] Saab and Dudley 1998.
[e] Martin and Li 1992.
[f] Saab et al., unpubl. data.
[g] Goggans et al. 1988. Values are apparent nest success.
[h] Dixon and Saab 2000. Values are apparent nest success.
[i] Garrett et al. 1996 (two regions reported separately). Values are apparent nest success.
[j] Germaine and Germaine 2002 (two treatments reported separately).
[k] Purcell et al. 1997.

are best for estimating abundances of songbirds but usually underestimate those species that do not sing consistently and those with large home ranges (e.g., woodpeckers and raptors) (cf. Martin and Eadie 1999). Results for the groups that may be under estimated should be treated with caution and are likely biased toward non-detection.

While considerable differences exist among these three studies, some patterns do emerge. Several species were consistently present in recently burned forests (e.g., Three-toed Woodpecker [*Picoides tridactylus*], Black-backed Woodpecker, Olive-sided Flycatcher [*Contopus cooperi*], Mountain Bluebird), whereas others were consistently more abundant in unburned forests (e.g., Golden-crowned Kinglet, Mountain Chickadee, Hermit Thrush). The majority of species showed mixed or no response across studies. These species are likely affected by fire-related factors including burn severity, time since fire, and total burn area (Kotliar et al. 2002). Typical survey techniques (i.e., point counts) likely cannot detect such effects without more comprehensive study design and replication.

No studies followed bird responses from early to late postfire stages. Results from Hutto (1995) and Harris (1982) are snapshots of bird species composition in early postfire years (1–4 yr postfire). Kotliar et al. (2002) examined forests for 8 yr postfire but did not estimate abundance or density of species encountered, so changes in species responses during the study are unknown. Regrowth of understory vegetation and associated increases of free-flying arthropods, loss of residual snags, and decline of bark and wood-boring beetles can dramatically change bird species composition of burned forests in later successional stages (e.g., >5–10 yr postfire). Long-term studies that follow burned forests through these successional stages are needed (e.g., Saab et al. 2004).

Several studies have noted an increase in cavity-nesting bird densities up to 3–5 yr postfire (Taylor and Barmore 1980, Caton 1996, Hitchcox 1996, Saab and Dudley 1998). Harris (1982) noted an increase in secondary cavity-nesting bird species but a decline in woodpecker densities 3 yr postfire. Such declines may be a response to decreases in bark and wood-boring beetles with increasing year postfire (Harris 1982, Dixon and Saab 2000, Powell 2000).

Abundance may not reflect population status without corresponding information on reproductive success (Brawn and Robinson 1996, Bock and Jones 2004). We know of only one study that examined nest success in burned mixed coniferous forests of the Rocky Mountains. Hitchcox (1996) compared nesting densities and success of cavity-nesting birds in salvage-logged and unlogged burned forests of northwestern Montana 2–4 yr postfire. Hitchcox selected seven salvage-logged plots 7–34 ha in size, in which most large trees (>15 cm diameter, >4.5 m tall) were removed. Densities of cavity nests were two to three times higher in unlogged (18 nesting species) compared to salvage-logged plots (eight nesting species). Mayfield nest success for the three most abundant species was higher in unlogged than salvage-logged treatments for Northern Flicker (95% vs. 67%, respectively, both N = 24 nests) and Mountain Bluebird (67%, N = 25 vs. 34%, N = 15) and similar between treatments for House Wren (*Troglodytes aedon*) (73%, N = 34 vs. 80%, N = 9; Hitchcox 1996).

The varied responses to fire by birds associated with mixed coniferous forests reflects the mixed fire regimes characteristic of these forests, and indicates a need for both understory and stand-replacement fires (Schoennagel et al. 2004). A return to frequent understory fire in lower elevations and rare stand-replacement fire at higher elevations would provide habitat for the diverse bird communities using mixed coniferous forests. Fire exclusion or management using only prescribed fire would not provide the mosaic of habitat conditions necessary to maintain the variation in avian communities associated with these forests.

LODGEPOLE PINE FORESTS

Lodgepole pine forests of the Rocky Mountains occur at middle to high elevations in the subalpine zone. These forests typically burn infrequently and at high severity (Schoennagel et al. 2004), although at lower elevations, small surface fires occasionally burn (Kipfmueller and Baker 2000).

Lodgepole pine is shade intolerant with few lateral branches, but tends to grow in very dense stands. Over time the dense stands naturally thin, contributing to abundant dead ladder fuels (Schoennagel et al. 2004). These abundant fuels, the low crown height, and the sparse surface fuels all promote high-severity crown fires. Severe drought and strong winds are necessary for fire to spread through the wetter fuels of subalpine forests. Typically, it takes decades or centuries for appropriate fuel accumulation and climatic conditions to coincide (Romme and Knight 1981). The lower fire-return intervals probably average from 60–80 yr (Agee 1993) and the upper return intervals from 100 to >500 yr (Romme and Knight 1981).

No evidence suggests that fire suppression has changed lodgepole stand structures in recent decades (Schoennagel et al. 2004). Fire histories demonstrate that long fire-free periods (as long as or longer than the fire exclusion period during the twentieth century) characterized the fire regimes of these forests prior to European settlement (e.g., Romme 1982, Veblen 2000).

BIRDS OF LODGEPOLE PINE FORESTS

No bird species are restricted to lodgepole pine forests, yet many use this habitat for some portion of their life history. Some species that use lodgepole pine forests include Spruce Grouse (*Falcipennis canadensis*), Three-toed Woodpecker, Clark's Nutcracker (*Nucifraga columbiana*), Ruby-crowned Kinglet, and Pine Siskin (Hein 1980, Hutto and Young 1999). Partners in Flight lists no priority species for this habitat, although several species that use lodgepole forests are priority species in other habitats (Partners in Flight 2004).

BIRD RESPONSE TO FIRE IN LODGEPOLE PINE FORESTS

Several studies have examined bird response to fire in lodgepole pine forests by comparing burned and unburned habitats (Davis 1976, Pfister 1980, Taylor and Barmore 1980, Caton 1996, Hoffman 1997). Most of these studies measured bird response as abundance or density estimates based on strip transect surveys (Davis 1976, Taylor and Barmore 1980), a combination of line transect surveys and spot-mapping (Pfister 1980), or fixed-radius point counts (Caton 1996). Caton (1996) and Hoffman (1997) also compared cavity-nest abundances or densities in burned and unburned forests (Table 2).

While considerable differences in study design, habitat, and survey methods exist among these studies, some patterns emerged. As in mixed coniferous forests, certain species were always more abundant in burned forests (Black-backed Woodpecker, Three-toed Woodpecker, and Mountain Bluebird), whereas other species were more abundant in unburned forests (Mountain Chickadee, Golden-crowned Kinglet, and Hermit Thrush).

In lodgepole pine forests of south central Wyoming, Davis (1976) compared breeding bird densities and species richness in three treatments: (1) clearcut, (2) burned, and (3) unlogged, unburned, considered the control plots. Richness and density estimates of most species were highest in burned plots surveyed 5–10 yr postfire than in either clearcut or control plots. Pfister (1980) compared breeding bird densities in burned and unburned lodgepole pine and spruce-fir forests in Yellowstone National Park. In lodgepole forests, burned plots had higher species richness than unburned plots, and most species occurred at their highest densities at 4–5 yr postfire. Taylor and Barmore (1980) examined breeding bird densities in moderate-to-high severity burns of lodgepole pine forests 1–29 yr postfire and in mature forests that had not burned for at least 43 yr. Breeding bird densities were highest in forests 5–29 yr postfire. The authors suggested the closed canopy of lodgepole pine forests >40 yr old resulted in declines of bird densities. Wood-boring beetles were present within the first year postfire, followed by Three-toed and Black-backed Woodpeckers during the second year postfire. Densities of woodpeckers declined with declining numbers of wood borers. Cavities created by these species as well as Hairy Woodpecker coincided with an increase of secondary cavity nesters up to 5 yr postfire, when non-excavators reached their highest densities.

Caton (1996) estimated abundances of cavity nests in burned forests 2–6 yr after fire and compared these abundances to those reported for the same study area before it burned (McClelland 1977). Overall abundances were higher in burned forests, although nests for some species (Red-naped Sapsucker [*Sphyrapicus nuchalis*], Red-breasted Nuthatch [*Sitta canadensis*], and chickadees) were more abundant in unburned forests. Bird abundance data obtained from point counts showed a positive response to fire by wood drillers, aerial insectivores, and ground foragers, whereas foliage and bark gleaners were less abundant in burned forests. Caton (1996) also found lower densities of cavity nests in salvage-logged compared to unlogged burned forests. Relative abundance of tree-foraging species was significantly lower in salvage-logged plots, whereas non-tree foraging species showed mixed responses.

Hoffman (1997) compared nest distributions of Three-toed, Black-backed, and Hairy Woodpeckers among three forest conditions: (1) burned, unlogged forest (2) unburned, clearcut forest and (3) unburned, mature lodgepole pine forest, termed undisturbed forest. Nests of all three species were over five times more likely to be found in 1-yr-old burned forests than in undisturbed forests. Nests of all three species were over 17 times more likely to be found in burned forests 2 yr postfire than in clearcuts.

Birds of lodgepole pine forests need little in the way of new fire management practices because fire regimes in these forests have seen little alteration since European settlement. Large stand-replacement fires are necessary for biological diversity in lodgepole pine forests (Agee 1993, Arno 2000). Infrequent, stand-replacement fires in this forest type clearly favor many bird species, especially cavity-nesting birds and flycatchers (Table 2). Stand-replacement fire regimes can be controlled by creating fuel breaks near property boundaries to protect resorts and other private facilities (see Arno 2000). This practice is likely to have few impacts on lodgepole pine bird communities if conducted on small spatial scales.

SPRUCE-FIR FORESTS

Spruce-fir forests occur at the highest forested elevations in the Rocky Mountains. Engelmann spruce (*Picea engelmannii*) and subalpine fir (*Abies lasiocarpa*) are the dominant trees. Whitebark pine (*Pinus albicaulis*) grows in drier regions. Infrequent, high-severity crown fires generally occur at intervals of 100 to >500 yr (Romme and Knight 1981). Successive seasons of drought can initiate large, stand-replacing fires in these typically moist forests (Balling et al. 1992). Drought-induced large fires are very rare but account for the greatest area burned in subalpine forests (Bessie and Johnson 1995). Similar to lodgepole pine, the spruce-fir forest floor lacks fine fuels, which propagate understory fires, on the forest floor. Rather, these dense forests have abundant ladder fuels that carry fire into tree crowns (Schoennagel et al 2004).

Efforts to suppress fires in systems with long-fire-return intervals have had limited success (Romme and Despain 1989, Schoennagel et al. 2004). Variation in climate rather than fuels appears to have the greatest influence on the size, timing, and severity of fires in spruce-fir and other subalpine forests (Romme and Despain 1989, Rollins et al. 2002, Schoennagel et al. 2004).

BIRDS OF SPRUCE-FIR FORESTS

Many species that occur in mixed coniferous and lodgepole pine forests also occur in spruce-fir forests. Some species that are consistently found in spruce-fir forests throughout the Rocky Mountains include Clark's Nutcracker, Ruby-crowned Kinglet, Hermit Thrush, Yellow-rumped Warbler, Pine Grosbeak, Red Crossbill (*Loxia curvirostra*), and Pine Siskin (Smith 1980). Partners in Flight lists no priority species specifically for this habitat, although several species that use spruce-fir forests are priority species in other habitats (Partners in Flight 2004).

BIRD RESPONSE TO FIRE IN SPRUCE-FIR FORESTS

We know of two studies that measured bird responses to wildland fire in spruce-fir forests (Pfister 1980, Taylor and Barmore 1980). In both studies, species richness and composition were similar between stand-replacement burns and unburned spruce-fir forests. Breeding bird densities, however, were higher in 2–3 yr old burned forests compared to unburned forests (Pfister 1980). Although Taylor and Barmore (1980) reported similar breeding bird densities between burned forests (1–3 yr after fire) and unburned forests, densities of Three-toed, Black-backed, and Hairy woodpeckers were higher in moderately burned forests.

Studies of burned and unburned spruce-fir forests report relatively minor differences in bird communities. Still, there is a clear pattern for some woodpecker species to favor burned habitats. Similar to lodgepole pine forests, alterations in historical fire regimes have been inconsequential for spruce-fir forests. Habitats created by rare, stand-replacing fire are characteristic of these high-elevation forests and necessary for the long-term persistence of the associated bird communities. Fire suppression is generally difficult and likely does not threaten the natural fire regimes or associated bird communities due to the remote nature of this habitat.

MANAGEMENT IMPLICATIONS AND RESEARCH QUESTIONS

After reviewing the literature on birds and fire in the Rocky Mountains, we suggest that the following six areas are highly deserving of management and research attention.

1. Future research should focus on the influence of burn severity and patch size to refine our understanding of how birds respond to fires. Severity and patch size could be incorporated into the response classes of Kotliar et al. (2002). We believe that groups of bird species can be identified that respond similarly to fires of certain severities or sizes. First approaches might be best aimed at distinguishing responses to low vs. high severity and large vs. small patches. Eventually this research could greatly improve our understanding of the mixed severity fires that govern many of the forests in the Rocky Mountains.
2. Long-term studies (at least 10 yr) are needed to explain postfire changes in habitats and avifauna. Most studies of postfire bird communities end less than 5 yr postfire, even though descriptive accounts suggest that there is a characteristic avifauna of middle-successional forests (Hutto 1995). A few long-term studies are ongoing (i.e., Saab et al. 2004), but more are urgently needed to capture the variability that we know exists among forest types and fire regimes.

3. Avian responses to fire vary not only with severity, patch size, and time since fire, but also with landscape context of burns, and postfire salvage logging. Over the last two decades, postfire salvage logging has become increasingly prevalent and is often implemented with little regard for wildlife (e.g., Caton 1996, Saab et al. 2002). Many cavity-nesting birds are associated with burned forests, including woodpeckers designated as sensitive species by state and federal agencies. These woodpecker species respond variably to postfire salvage logging (Saab and Dudley 1998). Litigation on management of these species has impeded the implementation of postfire management activities. Thus, design criteria for postfire salvage logging that maintains nesting habitat for woodpeckers is needed for planning and implementation of postfire management actions (Saab et al. 2002).
4. Studies of bird relationships to fire have focused on species composition and abundance in stand-replacing wildfires. For an improved understanding of the ecological consequences of fire management for birds, more research is needed to examine reproductive success and other demographic parameters to evaluate the habitat quality and source/sink status of fire-created (prescribed and wildfire) and fire-excluded habitats.
5. Recently burned forests potentially function as ephemeral source habitats for several avian species, particularly cavity-nesting birds. Early postfire habitats provide increases in snags that offer greater opportunities for nesting and foraging (e.g., Hutto 1995), and a reduced risk of nest predation compared to unburned forests (Saab and Vierling 2001). In this summary, data reported for selected woodpecker species suggest a pattern of higher nest survival in burned than in unburned forests. High reproductive success and increased productivity in recently burned forests might be explained in part by a reduction or elimination of nest predators following stand-replacement fires (Saab and Vierling 2001). Recolonization of small mammalian and reptilian nest predators into forests affected by wildfire may take several years, thus predation rates are expected to be lower in the years immediately following fire (Saab and Vierling 2001, Saab et al. 2004). The predator-release benefit of burns is still hypothetical and needs to be tested.
6. Perhaps the most difficult question facing managers in this region is how to burn higher elevation forests that did not evolve with low-severity fire. Traditional low-severity prescribed fire is not likely to replicate historic stand conditions or avifauna in these forests, which include higher-elevation mixed coniferous forests and all lodgepole pine and spruce-fir forests (i.e., the majority of forest types in the Rocky Mountains). Research in recent large fires across the Rocky Mountains indicates that large burns support diverse and productive avifauna (Saab et al. 2004). Clearly, management of the disparate forests of this region requires both prescribed fire and wildland fire.

Managers are increasingly using prescribed fire and thinning to reduce fire severity. Birds will likely respond differently depending on cover types and size and severity of treatments. Therefore, managers should consider targeting a variety of stand densities that reflect historic variation (e.g., Ehle and Baker 2003). This approach calls for cooperation between managers and researchers to implement replicated experiments done at appropriate scales that rigorously assess the effects of different prescriptions on habitats and populations of birds. Strategies for fire management should not only reduce fire risk but also maintain habitat for avifauna and other components of biodiversity in the Rocky Mountains.

The limited applicability of the Southwest ponderosa pine paradigm, coupled with our limited understanding of fire history and fire effects on natural resources other than trees, suggests that large-scale forest restoration could pose significant ecological risks unless it is carefully targeted to move the structure, function, and disturbance of a system back to historical conditions suitable for that system. Prudent study and application of locally appropriate fire regimes will be key to maintaining diverse ecosystems (Landres et al. 1999, Allen et al. 2002). If we do not soften the pervasive view of forests as static and perpetually green, ecosystem restoration cannot be successful. Management that targets the full range of natural variability (up to and including crown fires) will be more successful and more cost effective than aiming for conditions inappropriate to local systems (Landres et al. 1999).

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